

Vulnerability and Responses of American Samoa  
Mangroves to Relative Sea-Level Rise

and

Pacific Island Region Capacity-Building Priorities

By

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Doctor of Philosophy



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## SUPPORTING PUBLICATIONS AND REPORTS

Some of the work presented in this thesis appears in peer-reviewed publications and technical reports. Where substantial parts of these publications and reports are reproduced in this thesis, it is in all cases my own contributions to those publications and reports that are transposed. The publications and reports directly related to the work undertaken for this thesis are listed below.

- Gilman, E., J. Ellison, N. Duke, C. Field. 2008. Review. Threats to mangroves from climate change and adaptation options. *Aquatic Botany*. 89: 237-250.
- Gilman, E. J. Ellison. 2007. Efficacy of alternative low-cost approaches to mangrove restoration, American Samoa. *Estuaries and Coasts* 30(4): 641-651.
- Gilman, E., J. Ellison, R. Coleman. 2007. Assessment of mangrove responses to projected relative sea-level rise and recent historical reconstruction of shoreline position. *Environmental Monitoring and Assessment* 124(1-3): 105-130.
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- Gilman, E., J. Ellison, V. Jungblat, H. VanLavieren, L. Wilson, F. Areki, G. Brighthouse, J. Bungitak, E. Dus, M. Henry, I. Sauni Jr., M. Kilman, E. Matthews, N. Teariki-Ruatu, S. Tukia, K. Yuknavage. 2006. Capacity of Pacific island countries and territories to assess vulnerability and adapt to mangrove responses to sea-level and climate change. *Climate Research* 32(3): 161-176.
- Gilman, E., H. Van Lavieren, J. Ellison, V. Jungblut, L. Wilson, F. Areki, G. Brighthouse, J. Bungitak, E. Dus, M. Henry, M. Kilman, E. Matthews, I. Sauni Jr., N. Teariki-Ruatu, S. Tukia, K. Yuknavage. 2006. *Pacific Island Mangroves in a Changing Climate and Rising Sea*. UNEP Regional Seas Reports and Studies No. 179. United Nations Environment Programme, Regional Seas Programme, Nairobi, KENYA. ISBN 92-807-2741-9. ([www.unep.org/PDF/mangrove-report.pdf](http://www.unep.org/PDF/mangrove-report.pdf)).
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*Australia*. ISBN 1-934061-03-4. Western Pacific Regional Fishery Management Council and United Nations Environment Programme Regional Seas Programme, Honolulu, USA and Nairobi, Kenya, pp. 18-41.

Gilman, E. 2004. *Assessing and Managing Shoreline Responses to Projected Relative Sea-level Rise and Climate Change*. Prepared for the UNESCO World Heritage Central Pacific Project Planning Workshop, 5-9 October 2004, Kiritimati Island, Republic of Kiribati.

## ABSTRACT

An assessment was made of American Samoa mangroves' vulnerability and predicted changes in position from sea-level rise. The study also evaluated capacity in the Pacific Islands region to assess mangrove vulnerability to climate change and institute adaptation measures. Of the outcomes from climate change, relative sea-level rise may be the greatest threat to mangroves. By 2100, mangrove losses from relative sea-level rise could be as high as 47 percent in American Samoa and 22 percent when extrapolating regionally, causing about a quarter of total predicted annual regional losses.

American Samoa mangrove vulnerability to sea-level rise was determined and future position was predicted through analyses of sea-level trends and projections, mangrove spatial change analysis, reconstruction and monitoring of sedimentation rates, and determination of potential migration areas. These analyses provided three categories of requisite information: (i) Observed and projected rate of change in sea-level relative to the mangrove sediment surface, determined from trends in relative sea-level through analysis of sea-level data from a local tide gauge and observations of trends in the elevation of mangrove sediment surfaces; (ii) observed and projected trends in mangrove seaward margin positions; and (iii) physiographic settings (slope of land upslope and location of obstacles along the landward margin).

Results indicate that American Samoa mangroves are not likely keeping pace with rising sea-level, both surface and subsurface process controls on sediment elevation are important factors, and a large proportion (16, 23 and 68 percent) of the landward margins of the three mangrove study sites are obstructed from natural landward migration with sea-level rise. Based on observed trends in sediment surface elevations and movement of two mangroves' seaward margins, these sites have likely not been keeping pace with relative sea-level rise, with an elevation deficit of about 2 mm a<sup>-1</sup> at both sites. An embayment mangrove experienced sea-level rise relative to the mangrove sediment surface of 2.22 ( $\pm$  2.22 95% CI) mm a<sup>-1</sup> and a basin mangrove experienced 1.97 ( $\pm$  0.32 95% CI) mm a<sup>-1</sup>. At these sites, a highly significant positive correlation between the change in position of the seaward margins and change in relative sea-level suggests that rising sea-level

relative to the mangrove surface caused the observed landward migration. Shoreline movement was not significantly correlated with changing sea-level at a third site, where development activities have likely been dominant factors determining changes in mangrove position; vulnerability, based on observations of trends in sediment surface elevation, was not determined for this third site.

This study was the first to employ broad spatial coverage and a large number of sampling locations (330 sampling locations) to observe trends in the elevation of two mangroves' sediment surfaces, a necessary sampling design to adequately characterize mangrove sites, based upon previously documented high spatial variability in trends in mangroves' surface elevation. Both surface and subsurface processes exhibited large controls on sediment elevation, highlighting the need to monitor the full soil profile to accurately measure trends in mangrove surface elevation. Highly significant different mean changes in sediment surface elevation occurred for mangroves in different geomorphic settings (a difference of  $3.4 \pm 1.3 \text{ SE} \text{ mm a}^{-1}$ ,  $N = 1412$ ,  $P < 0.007$ ), supporting the hypothesis that mangroves in an estuarine/drowned river valley composite geomorphic setting are more resistant to relative sea-level rise than embayment mangroves. Mean landward migration of the mangroves' seaward margins was 12 to 37 times the relative sea-level rise rate. This is the first documentation of significantly different mean sediment surface elevation change for mangroves in different geomorphic settings, and the first documentation of the relationship between the rate of seaward mangrove margin erosion and relative sea-level rise rate, information needed to develop reliable predictive elevation models for mangrove ecosystems. Changes in extreme high water levels and frequency were found to not pose an increasing threat to American Samoa mangroves beyond the effects from rising mean sea-level. This site-specific assessment supports the hypothesis that, in this region, which experiences large El Nino Southern Oscillation-related steric changes lasting several months to years, extreme high waters are likely to be related to mean sea-levels.

This was the first comprehensive assessment to determine both (i) whether the mangrove site's threshold for resistance to sea-level rise has been exceeded, and (ii) the site's capacity to naturally migrate landward in response to rising sea-level. This was the first study to select research

methods suitable for employment in Pacific Small Island Developing States, considering both cost and staff abilities.

Results support instituting adaptation measures in American Samoa to reduce obstacles to landward mangrove migration with sea-level rise and to manage activities within catchments that affect mangrove elevation.

Regionally, there is extremely low capacity to assess mangrove vulnerability to climate change and to institute adaptation measures. Regional adaptation priorities include coastal planning that facilitates mangrove migration with sea-level rise, better management of non-climate stressors, and identification of climate change impacts on mangroves through regional standardized monitoring.

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Extremely helpful comments were provided by six anonymous peer reviewers of three manuscripts, which are based on the research presented in this thesis. These articles have been published in the journals *Climate Research*, *Environmental Monitoring and Assessment*, and *Wetlands Ecology and Management*. A fourth manuscript, based on research presented in Appendix 1 on American Samoa mangrove restoration, has been published in the journal *Estuaries and Coasts*. Finally, a fifth manuscript, an invited contribution to a special issue of the journal *Aquatic Botany*, provides a review of the state of knowledge of the effects on mangrove ecosystems from predicted climate changes, including relative sea-level rise, and considers adaptation options.

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# Chapter 1

## Introduction

### 1.1. THREATS TO MANGROVES FROM CLIMATE CHANGE

Mangrove ecosystems are threatened by outcomes from human-induced changes in the atmosphere's composition and alterations to land surfaces, including changes in sea-level, high water events, storminess, precipitation, surface temperature, atmospheric CO<sub>2</sub> concentration, ocean circulation patterns, health of functionally linked neighboring ecosystems, as well as human responses to climate change. Of all these outcomes, sea-level rise may be the greatest threat to mangrove ecosystems (Field, 1995; Lovelock and Ellison, 2007). Although, to date, it has likely been a smaller threat than anthropogenic activities such as conversion for aquaculture and filling (IUCN, 1989; Primavera, 1997; Ramsar Secretariat, 1999; Smith et al., 2001; Valiela et al., 2001; Alongi, 2002), relative sea-level rise is a substantial cause of recent and predicted future reductions in the area and health of mangroves and other tidal wetlands (IUCN, 1989; Ellison and Stoddart, 1991; Brown, 1997; Hubbard, 1997; Nichols et al., 1999; Ellison, 2000; McLean et al., 2001; Nurse et al., 2001; Cahoon and Hensel, 2006; McLeod and Salm, 2006). Rising sea-level will have the greatest impact on mangroves experiencing net lowering in elevation of the sediment surface that are in a physiographic setting that provides limited area for landward migration due to obstacles or steep gradients.

A suite of site-specific factors determine a mangrove's resistance and resilience to relative sea-level rise, including the geomorphic and physiographic setting, surface and subsurface controls on sediment surface elevation, effects from other stressors, and interactions and feedbacks between these factors (Woodroffe, 2002; Cahoon et al., 2006). Resistance is used here to refer to a mangrove's ability to keep pace with rising sea-level without alteration to its functions, processes and structure (Odum, 1989; Bennett et al., 2005). Resilience refers to the capacity of a mangrove to naturally migrate landward in response to rising sea-level, such that the

mangrove ecosystem absorbs and reorganizes from the effects of the stress to maintain its functions, processes and structure (Carpenter et al., 2001; Nystrom and Folke, 2001). For mangrove sites experiencing a rise in sea-level relative to the elevation of the mangrove's sediment surface, the depth, frequency and duration of flooding (hydroperiod) will increase, and the ecosystem may transgress landward, where unobstructed. Mangrove ecosystems were able to persist through the Quaternary despite substantial disruptions from large sea-level fluctuations, demonstrating that mangroves are highly resilient to change over historic time scales (Woodroffe, 1987, 1992). However, over coming decades, mangrove responses will not only be a result of how sea level is changing relative to the mangrove sediment surface and the physiographic setting, but will also be highly influenced by anthropogenic disturbances.

Mangroves provide regional and site-specific ecosystem services and products (e.g., Lewis, 1992; Ewel et al., 1998). Accurate predictions of changes to mangrove area and health, including in response to projected relative sea-level rise and other climate change outcomes, enable planning appropriate for specific sections of coastline to minimize and offset anticipated losses (Titus, 1991; Mullane and Suzuki, 1997; Ramsar Bureau, 1998; Hansen and Biringer, 2003). Small island states have limited capacity to adapt to relative sea level rise, including accommodating landward migration of mangroves and other coastal ecosystems. This is a result of their small land mass, high population densities and population growth rates, limited funds, poorly developed infrastructure, and susceptibility to damage from natural disasters (Nurse et al., 2001). Therefore, conducting planning now for adaptation to coastal ecosystem responses to relative seas-level rise and other climate change outcomes, instead of waiting for problems to become critical, is particularly appropriate for small island developing states and territories.

## **1.2. MANGROVE ECOSYSTEM SERVICES**

The Pacific Islands contain roughly three percent of the world's mangrove area, a small area in global terms, but each island group has a unique mangrove community structure (Ellison, 2000) and mangroves perform

valued site-specific and regional ecosystem services (e.g., Lewis, 1992; Ewel et al., 1998; Gilman, 1998). For instance, reducing coastal erosion and storm energy is a site-specific valued function, while provision of habitat to migratory shorebirds is a regional valued function. Therefore, threats to mangroves from climate change are of high importance to the Pacific Islands. Pacific Island governments recognize the value of mangroves and the need to augment conservation efforts (e.g., South Pacific Regional Environment Programme, 1999a). Mangroves are valued by people in the Pacific Islands region, for their provision of numerous ecosystem services and products:

- **Cultural practices.** Mangroves support traditional activities conducted by Pacific Islanders (e.g., Thaman, 1992; Ellison, 2001). Mangroves are a source of (i) clams, crabs, fish, and Tahitian chestnuts (*Inocarpus fagifera*), which are collected for consumption; (ii) wood used for construction, handicrafts, and fuel; (iii) *Ceriops tagal* wood used as part of a wedding dowry in the Central Province of Papua New Guinea; (iv) materials used for fishing equipment; (v) dye from pigments in *Bruguiera gymnorhiza* mangrove bark used in tapa in Polynesia and dye in Rhizophoraceae mangrove bark used to treat textiles, nets, and fish traps owing to its fungicidal properties; (vi) thatch used for mats and roofs; and (vii) plants used to make traditional medicines, such as infusion of Tahitian chestnut bark to treat stomachaches.
- **Coastal protection.** Mangroves protect coastlines and development from erosion and damage by tidal surges, currents, rising sea level, storm energy in the form of waves, storm surges and wind, and tsunamis, as most recently observed following the 2004 Indian Ocean tsunami (Danielsen et al., 2005; Kathiresan and Rajendran, 2005; Dahdouh-Guebas et al., 2005, 2006). Roots bind and stabilize the substrate (e.g., Krauss et al., 2003). For coastlines where relative sea level is rising, protecting mangroves is one way to slow anticipated erosion. Protecting mangroves sustains natural protection, and is less expensive than seawalls and similar erosion control structures, which can increase erosion in front of the structure and on adjacent properties.
- **Wildlife habitat.** Mangroves are nursery habitat for many wildlife species, including commercial fish and crustaceans, and thus contribute to

sustaining local abundance of fish and shellfish populations (e.g., Lal et al., 1984; Ley et al., 2002). As fish grow and become less vulnerable to predators, they move from the protective mangrove environment to mudflats, seagrass beds and coral reefs where foraging efficiency increases due to changes in their diet (Laegdsgaard and Johnson, 2001; Mumby et al., 2004). While mangroves in the Caribbean have been demonstrated to support juvenile coral reef fish (Mumby et al., 2004), mangroves in Papua New Guinea and the Solomon Islands have been found to be important nurseries for sandy and muddy-bottom demersal and surface feeding species, and not coral reef species (Quinn and Kojis, 1985; Blaber and Milton, 1990). Many migratory species depend on mangroves for part of their seasonal migrations. For instance, an estimated two million migratory shorebirds of the East Asian-Australasian Flyway, which annually migrate from the Arctic Circle through Southeast Asia to Australia and New Zealand and back, stop to forage at numerous wetlands along this flyway, including the wetlands of Oceania (Environment Australia, 2000). In addition to shorebirds, other waterbirds (e.g., wading birds and waterfowl), some of which are widely dispersing, and others which are more stationary, have population dynamics that make them dependent on wetlands (e.g., Haig et al., 1998).

- **Coastal water quality.** Mangroves maintain coastal water quality by abiotic and biotic retention, removal, and cycling of nutrients, pollutants, and particulate matter from land-based sources, filtering these materials from water before they reach seaward coral reef and seagrass habitats (e.g., Ewel, 1997; Ewel et al., 1998; Victor et al., 2004). Mangrove root systems slow water flow, facilitating the deposition of sediment. Toxins and nutrients can be bound to sediment particles or within the molecular lattice of clay particles and are removed during sediment deposition. Chemical and biological processes may transform and store nutrients and toxins in the mangrove sediment and vegetation. Some wetland plants can concentrate heavy metals in their tissues up to 100,000 times the concentration in ambient waters, and many of these plants contain substances that bind heavy metals and are involved in metal detoxification (Davies and Claridge, 1993).

- **Functionally linked coastal ecosystems.** Mangroves are functionally linked to neighboring coastal ecosystems (Mumby et al., 2004). For instance, terrigenous sediments and nutrients carried by freshwater runoff are first filtered by coastal forests, then by mangrove wetlands, and finally by seagrass beds before reaching coral reefs. The existence and health of coral reefs are dependent on the buffering capacity of these shoreward ecosystems, which support the oligotrophic conditions needed by coral reefs to limit overgrowth by algae (Ellison, 2004; Victor et al., 2004). Coral reefs, in turn, buffer the soft sediment landward ecosystems from wave energy (Ellison, 2004). Mangroves supply nutrients to adjacent coral reef and seagrass communities, sustaining these habitats' primary production and general health (Alongi et al., 1992; Dittmar et al., 2006). The importance of mangroves to the secondary productivity of adjacent coastal ecosystems has been demonstrated through organic export studies and the development of mangrove energy budgets (Mitsch and Gosselink, 1993). Also, mangroves provide a natural sunscreen for coral reefs, reducing exposure to harmful solar radiation and risk of bleaching: decomposing phytoplankton detritus and decaying litter from mangroves and seagrass beds produce a colored, chromophoric component of dissolved organic matter, which absorbs solar ultraviolet radiation, which can be transported over adjacent coral reefs and reduce coral reef exposure to harmful solar radiation (Anderson et al., 2001; Obriant, 2003).
- **Carbon sink:** Mangroves are a carbon sink; mangrove destruction releases large quantities of stored carbon and exacerbates global warming and other climate change trends. Conversely, mangrove rehabilitation will increase the sequestering of carbon (Kauppi et al., 2001; Ramsar Secretariat, 2001; Chmura et al., 2003).
- **Recreation, tourism, education and research:** Mangroves provide recreational and tourism opportunities, such as boardwalks and boat tours, and are important for research and education.

Economic valuation of mangrove ecosystems needs to be treated with caution, as most cost-benefit analyses included in site planning only examine costs and benefits as measured by market prices, ignoring mangrove and

other coastal system values not described by established monetary indicators (Dixon and Sherman, 1990; Ramsar Bureau, 1998; Wells et al., 2006). For instance, cultural and aesthetic quality-of-life benefits derived from ecosystems are not easily assigned economic value. Furthermore, economic valuation of ecosystems can produce different results depending on the length of time being considered and whether or not future values, such as a mangroves' future potential for tourism, are considered, and other assumptions (Dixon and Sherman, 1990; Ramsar Bureau, 1998; Wells et al., 2006). Having clarified these limitations, economic valuation is useful, as having a dollar value on mangrove functions is often needed to convince decision-makers of the importance of mangrove benefits, and the concomitant need for and benefits of mangrove conservation (Ramsar Bureau, 1998; Wells et al., 2006).

The annual economic values of mangroves, estimated by the cost of the products and services they provide, have been estimated to be USD 200,000 -- 900,000 ha<sup>-1</sup> (Wells et al., 2006). The value of Malaysian mangroves just for storm protection and flood control has been estimated at USD 300,000 per km of coastline, which is based on the cost of replacing the mangroves with rock walls (Ramsar Secretariat, 2001). The mangroves of Moreton Bay, Australia, were valued in 1988 at USD 4,850 ha<sup>-1</sup> based only on the catch of marketable fish (Ramsar Secretariat, 2001). Mangroves can also be provided with an economic value based on the cost to replace the products and services that they provide, or the cost to restore or enhance mangroves that have been eliminated or degraded. The range of reported costs for mangrove restoration is USD 225 to USD 216,000 ha<sup>-1</sup>, not including the cost of the land (Lewis, 2005). In Thailand, restoring mangroves is costing USD 946 ha<sup>-1</sup> while the cost for protecting existing mangroves is only USD 189 ha<sup>-1</sup> (Ramsar Secretariat, 2001).

### **1.3. THREATS TO MANGROVE FROM OTHER CLIMATE CHANGE OUTCOMES**

Numerous climate change factors other than change in relative sea-level are predicted to affect mangroves. Rise in temperature and the direct effects of increased CO<sub>2</sub> levels are likely to increase mangrove productivity, change



the timing of flowering and fruiting, and expand the ranges of mangrove species into higher latitudes. Changes in precipitation and subsequent changes in aridity may affect the distribution of mangroves. However, these other outcomes of climate change are less certain. For instance, there is a lack of agreement in the scientific community regarding whether increases in greenhouse gases will result in either a more El Nino- or La Nina-like tropical Pacific, which affects a range of weather conditions, including storminess, precipitation and extreme high waters (McLean et al., 2001; Vecchi et al., 2008). Mangrove responses to changes in these other parameters are not well understood. The understanding of the synergistic effects of multiple climate change stressors and other anthropogenic and natural stressors on mangroves is also poor.

#### **1.3.1. Extreme High Water Events**

The frequency of extreme high water events of a given height relative to fixed benchmarks is projected to increase over coming decades as a result of the same atmospheric and oceanic factors that are causing global sea-level to rise, and possibly also as a result of other influences on extremes such as variations in regional climate, like phases of the El Nino Southern Oscillation and North Atlantic Oscillation, through change in storminess and resulting storm surges (Hunter, 2002; Woodworth and Blackman, 2004; Church et al., 2001, 2004b). There have been several studies of trends in frequency and elevation of extreme high water events (Bijl et al., 1999; D'Onofrio et al., 1999; Woodworth and Blackman, 2002, 2004; Church et al., 2004b). For example, an analysis of 99th percentiles of hourly sea level at 141 globally distributed stations for recent decades showed that there has been an increase in extreme high sea level worldwide since 1975 (Woodworth and Blackman, 2004).

Church et al. (2006) observed a significant increase in monthly sea-level variance after 1970 in the Pacific region, indicating that there has been an increasing trend in inter-decadal variability, such as from ENSO events. This implies that, regionally, an increase in the frequency of extreme events has been occurring, and may become an increasingly prominent management issue. In many cases, the secular changes in extremes were found to be similar to those in mean sea level.

Increased frequency and levels of extreme high water events could affect the position and health of coastal ecosystems and pose a hazard to coastal development and human safety. Increased levels and frequency of extreme high water events may affect the position and health of mangroves by altering salinity, recruitment and inundation, in addition to changing the wetland sediment budget. For instance, increased levels and frequency of extreme high water events may affect the position and health of mangroves by altering salinity, recruitment and inundation, in addition to changing the wetland sediment budget, however, the state of knowledge of ecosystem effects from changes in extreme waters is poor.

### **1.3.2. Storms**

During the 21<sup>st</sup> century, the Intergovernmental Panel on Climate Change projects that there is likely to be an increase in tropical cyclone peak wind intensities and an increase in tropical cyclone mean and peak precipitation intensities in some areas as a result of global climate change (Houghton et al., 2001; Solomon et al., 2007). Storm surge heights are also predicted to increase if the frequency of strong winds and low pressures increase. This may occur if storms become more frequent or severe as a result of climate change (Church et al., 2001; Houghton et al., 2001; Solomon et al., 2007).

The increased intensity and frequency of storms has the potential to increase damage to mangroves through defoliation and tree mortality. In addition to causing tree mortality, stress, and sulphide soil toxicity, storms can alter mangrove sediment elevation through soil erosion, soil deposition, peat collapse, and soil compression (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Smith et al., 1994; Mastaller, 1996; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Baldwin et al., 2001; Sherman et al., 2001; Woodroffe, 1995b, 2002; Cahoon et al., 2003, 2006; Cahoon, 2006; Piou et al., 2006). Areas suffering mass tree mortality with little survival of saplings and trees might experience permanent ecosystem conversion, as recovery through seedling recruitment might not occur due to the change in sediment elevation and concomitant change in hydrology (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Woodroffe, 1995b, 2002; Cahoon et al., 2003). Other

natural hazards, such as tsunami, which will not be affected by climate change, can also cause severe damage to mangroves and other coastal ecosystems (e.g., the 26 December 2004 Indian Ocean tsunami [Danielsen et al., 2005; Kathiresan and Rajendran, 2005; Dahdouh-Guebas et al., 2005, 2006]).

### **1.3.3. Precipitation**

Globally, rainfall is predicted to increase by about 25% by 2050 in response to climate change. However, the regional distribution of rainfall will be uneven (Houghton et al., 2001). Increased precipitation is very likely in high-latitudes, and decreased precipitation is likely in most subtropical regions, especially at the poleward margins of the subtropics (Solomon et al., 2007). In the most recent assessment, the Intergovernmental Panel on Climate Change reported significant increases in precipitation in eastern parts of North and South America, northern Europe and northern and central Asia, with drying in the Sahel, the Mediterranean, southern Africa and parts of southern Asia (Solomon et al., 2007). Long-term trends had not been observed for other regions.

Changes in precipitation patterns are expected to affect mangrove growth and spatial distribution (Field, 1995; Ellison, 2000). Based primarily on links observed between mangrove habitat condition and rainfall trends (Field, 1995; Duke et al., 1998), decreased rainfall and increased evaporation will increase salinity, decreasing net primary productivity, growth and seedling survival, altering competition between mangrove species, decreasing the diversity of mangrove zones, causing a notable reduction in mangrove area due to the conversion of upper tidal zones to hypersaline flats. Areas with decreased precipitation will have a smaller water input to groundwater and less freshwater surface water input to mangroves, increasing salinity. As soil salinity increases, mangrove trees will have increased tissue salt levels and concomitant decreased water availability, which reduces productivity (Field, 1995). Increased salinity will increase the availability of sulfate in seawater, which would increase anaerobic decomposition of peat, increasing the mangrove's vulnerability to any rise in relative sea level (Snedaker, 1993, 1995). Reduced precipitation can result

in mangrove encroachment into salt marsh and freshwater wetlands (Saintilan and Wilton, 2001; Rogers et al., 2005).

Increased rainfall will result in increased growth rates and biodiversity, increased diversity of mangrove zones, and an increase in mangrove area, with the colonization of previously unvegetated areas of the landward fringe within the tidal wetland zone (Field, 1995; Duke et al., 1998). For instance, mangroves tend to be taller and more diverse on high rainfall shorelines relative to low rainfall shorelines, as observed in most global locations, including Australia (Duke et al., 1998). Areas with higher rainfall have higher mangrove diversity and productivity probably due to higher supply of fluvial sediment and nutrients, as well as reduced exposure to sulfate and reduced salinity (McKee, 1993; Field, 1995; Ellison, 2000). Mangroves will likely increase peat production with increased freshwater inputs and concomitant reduced salinity due to decreased sulfate exposure (Snedaker, 1993, 1995).

These predicted responses are based on assessments from only a few areas and are currently untested in longitudinal studies at any single location. Further research is needed to confirm these hypotheses and to assess the broader significance of rainfall variability on mangroves.

#### **1.3.4. Temperature**

Between 1906 and 2005 the global average surface temperature has increased by  $0.74^{\circ}\text{C}$  ( $\pm 0.18^{\circ}\text{C}$ ) (Solomon et al., 2007). The linear warming trend of the last fifty years ( $0.13^{\circ}\text{C}$  per decade) is nearly twice that for the last 100 years. This rise in globally averaged temperatures since the mid-20<sup>th</sup> century is considered to be very likely due to the observed increase in anthropogenic greenhouse gas atmospheric concentrations (Solomon et al., 2007). The range in projections for the rise in global averaged surface temperatures from 1980-1999 to the end of the 21<sup>st</sup> century (2090-2099) is  $1.1$  to  $6.4^{\circ}\text{C}$  (Solomon et al., 2007). Mangroves reach a latitudinal limit at the  $16^{\circ}\text{C}$  isotherm for air temperature of the coldest month, and the margins of incidence of ground frost, where water temperatures do not exceed  $24^{\circ}\text{C}$  (Ellison, 2000). The optimum mangrove leaf temperature for photosynthesis is believed to be between  $28$  to  $32^{\circ}\text{C}$ , while photosynthesis ceases when leaf temperatures reach  $38 - 40^{\circ}\text{C}$  (Clough et al., 1982; Andrews et al.,

1984). Increased surface temperature is expected to affect mangroves by (Field, 1995; Ellison, 2000):

- (i) Changing species composition;
- (ii) Changing phenological patterns (e.g. timing of flowering and fruiting);
- (iii) Increasing mangrove productivity where temperature does not exceed an upper threshold; and
- (iv) Expanding mangrove ranges to higher latitudes where range is limited by temperature, but is not limited by other factors, including a supply of propagules and suitable physiographic conditions.

The frequency, duration and intensity of extreme cold events have been hypothesized to explain the current latitudinal limits of mangrove distribution (Woodroffe and Grindrod, 1991; Snedaker, 1995). However, the incidence of extreme cold events is not likely to be a factor limiting mangrove expansion to higher latitudes in response to increased surface temperature. The Intergovernmental Panel on Climate Change projects reduced extreme cold events (Solomon et al., 2007), in correlation with projected changes in average surface temperatures. For instance, Vavrus et al. (2006) predicted a 50-100% decline in the frequency of extreme cold air events during Northern Hemisphere winter in most areas, while Meehl et al. (2004) projected decreases in frost days in the extratropics, where the pattern of decreases will be determined by changes in atmospheric circulation.

#### **1.3.5. CO<sub>2</sub> Concentration**

The atmospheric concentration of CO<sub>2</sub> has increased 35% from a pre-industrial value, from 280 parts per million by volume (ppmv) in 1880 to 379 ppmv in 2005 (Solomon et al., 2007). In recent decades, CO<sub>2</sub> emissions have continued to increase: CO<sub>2</sub> emissions increased from an average of  $6.4 \pm 0.4 \text{ GtC a}^{-1}$  in the 1990s to  $7.2 \pm 0.3 \text{ GtC a}^{-1}$  in the period 2000 to 2005.

The effect of enhanced CO<sub>2</sub> on mangroves is poorly understood and there is a paucity of research in this area. A direct effect of elevated atmospheric CO<sub>2</sub> levels may be increased productivity of some mangrove species (Field, 1995; Ball et al., 1997). Mangrove metabolic responses to increased atmospheric CO<sub>2</sub> levels are likely to be increased growth rates (Farnsworth et al., 1996) and more efficient regulation of water loss (UNEP,

1994). For some mangrove species, the response to elevated CO<sub>2</sub> may be sufficient to induce substantial change of vegetation along natural salinity and aridity gradients. Ball et al. (1997) showed that doubled CO<sub>2</sub> had little effect on mangrove growth rates in hypersaline areas, and this may combine with reduced rainfall to create some stress. The greatest effect may be under low salinity conditions. Elevated CO<sub>2</sub> conditions may enhance the growth of mangroves when carbon gain is limited by evaporative demand at the leaves but not when it is limited by salinity at the roots. There is no evidence that elevated CO<sub>2</sub> will increase the range of salinities in which mangrove species can grow. The conclusion is that whatever growth enhancement may occur at salinities near the limits of tolerance of a species, it is unlikely to have a significant effect on ecological patterns (Ball et al., 1997). However, not all species may respond similarly, and other environmental factors, including temperature, salinity, nutrient levels and the hydrologic regime, may influence how a mangrove wetland responds to increased atmospheric CO<sub>2</sub> levels (Field, 1995).

#### **1.3.6. Ocean Circulation Patterns**

Key oceanic water masses are changing, however, the Intergovernmental Panel on Climate Change reports that at present, there is no clear evidence for ocean circulation change (Bindoff et al., 2007). There have been observations of long-term trends in changes in global and basin-scale ocean heat content and salinity, which are linked to changes in ocean circulation (Gregory et al., 2005; Bindoff et al., 2007).

Changes to ocean surface circulation patterns may affect mangrove propagule dispersal and the genetic structure of mangrove populations, with concomitant effects on mangrove community structure (Duke et al., 1998; Benzie, 1999; Lovelock and Ellison, 2007). Increasing gene flow between currently separated populations and increasing mangrove species diversity could increase mangrove resistance and resilience.

#### **1.3.7. Adjacent Ecosystem Responses**

Coral reefs, seagrass beds, estuaries, beaches, and coastal upland ecosystems may experience reduced area and health from climate change outcomes, including increased temperature, timing of seasonal temperature

changes, and ocean acidification (Harvell et al., 2002; Kleypas et al., 2006; Mydlarz et al., 2006). Mangroves are functionally linked to neighboring coastal ecosystems, including seagrass beds, coral reefs, and upland habitat, although the functional links are not fully understood (Mumby et al., 2004). Degradation of adjacent coastal ecosystems from climate change and other sources of stress may reduce mangrove health. For instance, mangroves of low islands and atolls, which receive a proportion of sediment supply from productive coral reefs, may suffer lower sedimentation rates and increased susceptibility to relative sea-level rise if coral reefs become less productive due to relative sea-level rise or other climate change outcomes. Mangroves receive sediment from coral reefs, where calcium carbonate is produced by corals, coralline algae, mollusks, forams, and echinoderms, which is made available from bioerosion and wave action, as well sediment production and reworking from coral sediment rejection and shedding, and a portion of loose sediment is transported landward coastal ecosystems (Hubbard and Miller, 1990; Glynn, 1996).

#### **1.3.8. Human Responses**

Anthropogenic responses to climate change have the potential to exacerbate the adverse effects of climate change on mangrove ecosystems. For instance, we can expect an increase in the construction of seawalls and other coastal erosion control structures adjacent to mangrove landward margins as the threat to development from rising sea-levels and concomitant coastal erosion becomes more apparent. As discussed previously, seawalls and other erosion control structures cause erosion and scouring of the mangrove immediately fronting and down-current from the structure (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997). Or, for example, areas experiencing reduced precipitation and rising temperature may have increased groundwater extraction to meet the demand for drinking water and irrigation. Increased groundwater extraction will increase sea-level rise rates relative to mangrove surfaces (Krauss et al., 2003), increasing mangrove vulnerability. Increased rainfall could lead to increased construction of stormwater drainage canals to reduce flooding of coastal upland areas, diverting surface water from mangroves and other coastal systems, reducing mangrove productivity.

#### 1.4. SEA-LEVEL RISE

Global eustatic sea-level rise is one of the more certain outcomes from changes in the atmosphere's composition and alterations to land surfaces. It is already likely taking place (12-22 cm of sea-level rise occurred during the 20th century), and several climate models as well as available sea-level data project an accelerated rate of global sea-level rise over coming decades (Cazenave and Nerem, 2004; Church et al., 2001, 2004a; Holgate and Woodworth, 2004; Thomas et al., 2004; Church and White, 2006; Bindoff et al., 2007; Solomon et al., 2007). The onset of the modern acceleration in global sea level rise started between the mid-19th and mid-20th centuries, in about 1870 (Solomon et al., 2007). Geological observations indicate that during the previous 2000 years, sea level change was small, with average rates in the range of  $0.0 - 0.2 \text{ mm a}^{-1}$  (Solomon et al., 2007). The observed rate of global sea level rise from 1961-2003, estimated from tide gauge data, is estimated to be  $1.8 \pm 0.5 \text{ mm a}^{-1}$ , whereas from 1993-2003, based on estimates from TOPEX/Poseidon satellite altimetry, the rate is  $3.1 \pm 0.7 \text{ mm a}^{-1}$ , which may reflect decadal variability or possibly acceleration in the rate of rise. For the 20<sup>th</sup> century, the estimated sea-level rise rate is  $1.7 \pm 0.5 \text{ mm a}^{-1}$  (Bindoff et al., 2007; Solomon et al., 2007). Recent findings on global acceleration in sea level rise indicate that the upper projections of the Intergovernmental Panel on Climate Change are likely to occur (Church and White, 2006). The range of projections for global sea-level rise from 1980-1999 to the end of the 21<sup>st</sup> century (2090-2099) is 0.18 – 0.59 m (Solomon et al., 2007).

'Relative sea level change', the change in sea level relative to the local land as measured at a tide gauge, is a combination of the change in eustatic (globally averaged) sea level and regional and local factors. The former is the change in sea level relative to a fixed Earth coordinate system, which, over human time scales, is due primarily to thermal expansion of seawater, changes in terrestrial water storage, and the transfer of ice from glaciers, ice sheets and ice caps to water in the oceans (Church et al., 2001; Hunter, 2002; Lambeck, 2004; Woodroffe and Horton, 2005; Bindoff et al., 2007; Solomon et al., 2007). The latter is the result of vertical motion of the land from tectonic movement, the glacio- or hydro-isostatic response of the Earth's crust to changes in the weight of overlying ice or water, coastal



subsidence such as due to extraction of subsurface groundwater or oil, geographical variation in thermal expansion, and for shorter time scales over years and shorter, meteorological and oceanographic factors, such as changes in density (from changes in temperature and salinity), winds from a constant direction, ocean circulation, and oceanographic processes such as El Nino phases and changes in offshore currents (Church et al., 2001; Hunter, 2002; Lambeck, 2004; Woodroffe and Horton, 2005; Bindoff et al., 2007; Solomon et al., 2007).

The rate of change of relative sea-level as measured at a tide gauge and fixed benchmarks may differ substantially from the relative sea-level rate of change occurring in coastal wetlands due to changing elevation of the wetland sediment surface, controlled by several processes (Section 1.5.4). Therefore, it is useful to distinguish between rates of change in relative sea-level at regional levels vs. the local level, such as within an individual coastal wetland ecosystem. In this thesis, "regional" relative sea-level rise or lowering is used to refer to how sea-level is changing on average for all of American Samoa. "Site specific" relative sea-level rise or lowering is used to refer to the change in sea-level relative to the elevation of the sediment surface of an individual coastal wetland site (Fig 1.1). The following equation and Fig. 1.1 elaborate on the difference and definition of regional vs. site-specific relative sea-level rise:

Equation 1. Rate of change in site-specific relative sea-level.

$$h_{rslm} = h_{rsltg} - h_{em}$$

where  $h_{rslm} = \Delta$  "site-specific relative sea-level" or  $\Delta$  in sea-level relative to the mangrove sediment surface

$h_{rsltg} = \Delta$  in sea-level relative to the local land, as measured at a tide gauge relative to a fixed benchmark

$h_{em} = \Delta$  in elevation of the mangrove sediment surface

For instance, if a tide gauge proximate to a mangrove has documented regional relative sea-level rise at a rate of  $2.0 \text{ mm a}^{-1}$ , and the mangrove sediment surface has been observed to be lowering at a rate of  $-1.0 \text{ mm a}^{-1}$ , then the site-specific relative sea-level rise rate is  $3 \text{ mm a}^{-1}$  (regional relative sea-level rise rate  $2.0 \text{ mm a}^{-1}$  minus rate of change in the sediment elevation

-1.0 mm a<sup>-1</sup>). In this example and that shown in Fig. 1.1, the rate of change in regional relative sea-level is positive while the rate of change in mangrove sediment elevation is negative, resulting in a rate of change in site-specific relative sea-level that exceeds the sea-level rise rate as measured by the local tide-gauge.

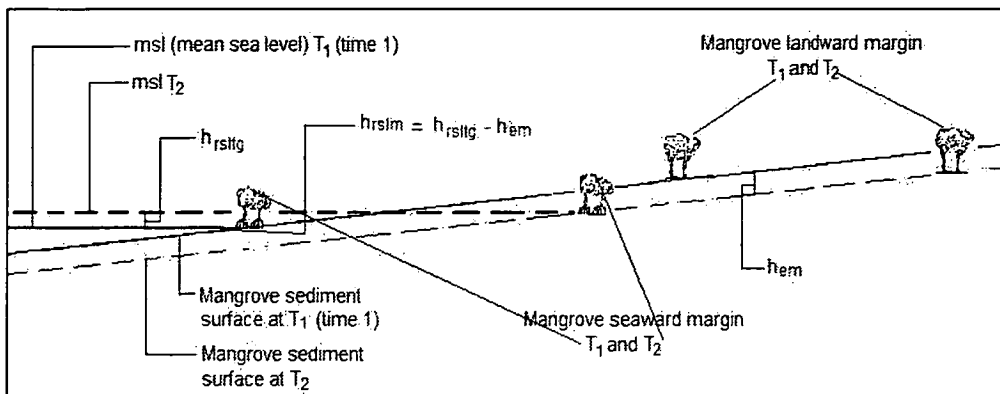


Fig. 1.1. Schematic demonstrating the changes in 'regional' and 'site-specific' relative sea-level, and change in mangrove sediment elevation.

To extract the signal of sea level change that is due to the volume of water in the global ocean, land motions need to be removed from tide gauge measurements using, for example, direct measurements of land motion using GPS and/or interpretations of dated paleoshoreline records (Fig. 1.2) (Farrell and Clark, 1976; Mitrovica and Milne, 2002). While land motions related to glacial isostatic adjustment (GIA) can be estimated with global geodynamic models (Lambeck, 2002, 2004; Peltier, 2001, 2002, 2004), estimating other land motions is usually not possible unless there are adequate and proximate geodetic or geological data, which is typically not the case (Bindoff et al., 2007). However, new techniques have been developed to attempt to overcome this constraint (Mitrovica and Milne, 2002). To determine projections in relative sea-level over coming decades, which is of interest due to its potential impact on mangroves and other coastal ecosystems (as well as coastal development and human populations), there is a need to understand how sea-level is changing relative to the site-specific elevation of the sediment surface of coastal ecosystems.

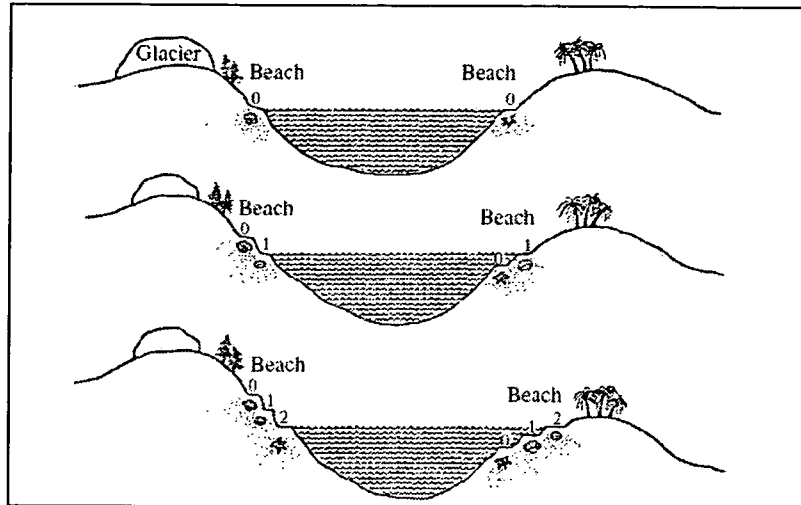


Fig. 1.2. Eustatic, regional, and local factors combine to determine site-specific records of past relative sea-level (Farrell and Clark, 1976). On the left-hand coastline, relative sea-level has fallen due to isostatic crustal rebound following partial deglaciation. The coastline on the right has experienced relative sea-level rise due to global eustatic sea-level rise resulting from loss of mass from the glacier in what might be a tectonically stable location, far from major glaciation centers.

#### 1.4.1. Human-Scale Influences on Global Sea-Level

Over human time scales of decades, the largest influences on global eustatic sea level are linked to climate and climate change processes (Intergovernmental Panel on Climate Change, 2001; Bindoff et al., 2007). Fig. 1.3 illustrates the processes that influence global sea-level, estimated contributions for the period 1910 to 1990, and the level of uncertainty for predicted influence on sea-level change rates. Global sea-level rise during the 20<sup>th</sup> century has very likely been significantly influenced by global warming through thermal expansion of seawater and loss of land ice (Intergovernmental Panel on Climate Change, 2001). Global ocean heat content has increased significantly since the late 1950s, more than two thirds of this increase in heat content has occurred in the upper 700 m of the ocean (Bindoff et al., 2007). Over the period 1961 to 2003, global ocean temperature has risen by 0.10°C from the surface to a depth of 700 m (Bindoff et al., 2007). The projected short-term sea-level rise from 1990 to 2100 is due primarily to thermal expansion of seawater and transfer of ice from glaciers and ice caps to water in the oceans, which both change the

volume of water in the world oceans (eustatic change) (Church et al, 2001; Lernke et al., 2007). Based on observational estimates, thermal expansion contributed about  $1 \text{ mm a}^{-1}$  to global sea-level rise over recent decades and contributed  $0.42 \pm 0.12 \text{ mm a}^{-1}$  for the period 1961-2003 (Church et al., 2001; Bindoff et al., 2007; Solomon et al., 2007). This steric expansion and contraction of the water column is primarily due to the effect of temperature and to a lesser degree, salinity (Woodroffe and Horton, 2005).

The increased melting of glaciers, ice caps and ice sheets is expected to have the second largest contribution to global sea level rise over the coming 100 years, which have contributed an estimated 0.2 to 0.4  $\text{mm a}^{-1}$  to sea-level rise over the 20<sup>th</sup> century, and contributed  $0.7 \pm 0.5 \text{ mm a}^{-1}$  for the period 1961-2003 (Church et al., 2001; Lambeck, 2002; Bindoff et al., 2007; Solomon et al., 2007). This increased melting and evaporation of glaciers and ice caps results in a change in the mass of water in world oceans, as the water in the ocean and volume of glacial ice on land are in balance, known as glacial eustasy (Woodroffe and Horton, 2005).

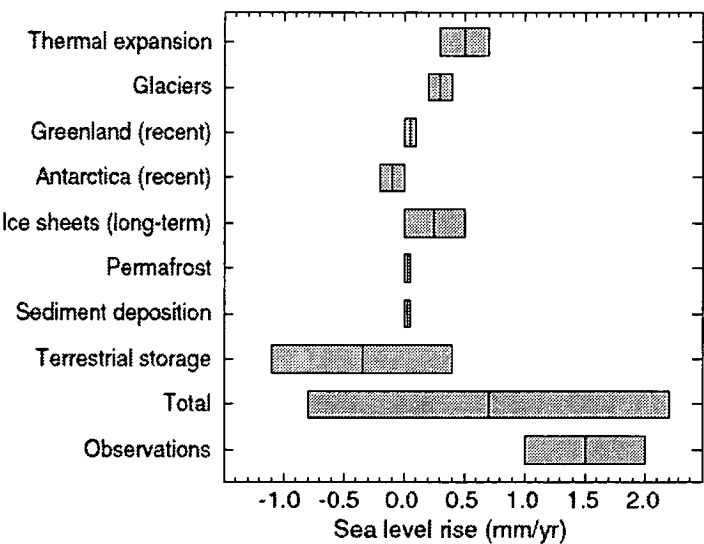


Fig. 1.3. Eight processes believed to contribute to the change in the rate of sea-level rise from 1910 to 1990, the estimated contribution from each process, and the range of uncertainty for each processes' contribution to average rate of sea-level rise during this period (Church et al., 2001).

The large ice sheets of Greenland and Antarctica could make a net contribution to sea level change in the coming century. Taken together, the

ice sheets in Greenland and Antarctica have very likely been contributing to sea level rise from 1993 to 2003. Thickening in central regions of Greenland has been more than offset by increased coastal melting (Lernke et al., 2007). Increased melting and evaporation on the Greenland and Antarctic ice sheets is projected but may be outweighed by increased precipitation as snow, and incorporation of most of this precipitation into the ice sheets (Church et al., 2001). The increased snowfall may be outweighed by accelerated ice flow and greater discharge, as recently observed in some marginal areas of the Antarctic ice sheets. The process of accelerated ice flow is not completely understood (Solomon et al., 2007). The melting of sea ice results in only a small eustatic sea-level change because the weight of sea ice is already afloat and supported by the ocean, and the consequent change in density and ocean volume caused by melting is a very small term (Noerdlinger and Brower, 2007). However, ice shelf changes can affect the flow of adjacent ice that is not floating, and thus affect sea level indirectly (Lernke et al., 2007).

Changes in terrestrial water storage will also alter the mass of the ocean, such as by groundwater extraction in excess of the rate of natural recharge (decreases the ocean water mass), impounding water in reservoirs above the water table (decreases the ocean water mass), reducing the volume of large lakes through irrigation and other water use (increases the ocean water mass), changes in the area of permafrost and an increased depth of the active layer (the soil layer of seasonally thawed ground above permafrost) (increases the ocean water mass), and deforestation (increases the ocean water mass). However, the effects of these processes on global sea-level change is not well understood. For instance, increased groundwater extraction and increased extraction of water from surface sources for irrigation may not affect sea-level if much of the extracted water infiltrates back into aquifers, but some of this water may reach the ocean through the atmosphere (evaporation and evapotranspiration) or surface runoff (Church et al., 2001; Woodroffe and Horton, 2005). Church et al. (2001) argues that the effect of changes in terrestrial water storage on sea level may be substantial, however, it is as yet unclear if the net effect will be to increase or decrease ocean water mass, as estimates of the change and effect of each factor are highly uncertain.

The change in ice volume since the Last Glacial Maximum has an effect on modern changes in global sea-level through GIA processes (Peltier, 1996, 2002, 2004; Mitrovica and Milne, 2002; Woodroffe and Horton, 2005). The transfer of crustal load from Pleistocene ice concentrated in the polar regions to ocean water widely distributed across the Earth's surface led not only to eustatic rise in global sea level but also to slow Holocene deformation of the Earth's mantle that caused relative sea level to fluctuate through time in various patterns in different regions (as ocean basins change their levels by crustal movements, this changes the total ocean volume, referred to as tectono-eustasy) (Dickinson and Green, 1998; Woodroffe and Horton, 2005). Gravitational equilibrium is restored following deglaciation by crustal 'rebound', and through the horizontal redistribution of water in the ocean basins, referred to as geoidal eustasy, required to maintain the ocean surface at gravitational equipotential (Woodroffe and Horton, 2005; Jansen et al., 2007). The vertical land movements induced by changes in surface loads of ice and water and concomitant redistribution of mass within the Earth and oceans, including the Earth's elastic and gravitational response to the changed water loading, takes thousands of years (Church et al., 2001; Woodroffe, 2002). Contribution of global GIA processes to the modern rate of global sea level rise has been measured by the TOPography EXperiment (TOPEX)/Poseidon (T/P) satellite to be  $-0.28 \text{ mm a}^{-1}$  for the ICE-4G(VM2) model of Peltier (1996) and  $-0.36 \text{ mm a}^{-1}$  for the ICE-5G(VM2) model of Peltier (2004).

Another contribution to global sea-level rise through 2100 is from deposition of sediment, which reduces ocean basin volume (Church et al., 2001). However, this is likely to have only a small contribution to sea-level change over short time periods, estimated to result in a sea-level rise rate of less than  $0.05 \text{ mm a}^{-1}$  (Church et al., 2001).

Mantle dynamic processes, which change the shape and volume of ocean basins, is expected to have a small influence on global sea-level over short time periods, estimated to result in a sea-level change rate of less than  $0.01 \text{ mm a}^{-1}$  (Church et al., 2001; Lambeck, 2004; Woodroffe and Horton, 2005).

The stability of the West Antarctic ice sheet, which is grounded below sea level and could possibly result in rapid ice discharge resulting in a global

sea-level rise of 6m if surrounding ice shelves are weakened, is unlikely to be lost and result in a substantial sea level rise during the 21st century. However, the understanding of its dynamics is not well understood, especially for longer time-scale projections (Church et al., 2001; Intergovernmental Panel on Climate Change, 2001; Lemke et al., 2007). Ice sheet models project that the West Antarctic ice sheet will contribute no more than 3mm a<sup>-1</sup> to sea-level rise over the next thousand years (Intergovernmental Panel on Climate Change, 2001).

#### **1.4.2. Glacial Cycle Time Scale Influences on Global Sea-Level**

Global climate change over geological, ice-age time scales, on the order of tens to hundreds of thousands of years, is primarily a result of changes in the input of incoming solar radiation and changes in the seasonal and latitudinal distribution of solar radiation, caused by Milankovitch astronomical cycles, and resulting glacial cycles (Hays et al., 1976; Pisias and Imbrie, 1986/1987; Komar, 1989; Lambeck, 2004; Solomon et al., 2007). These astronomical cycles consist of (i) change in the Earth's angle of tilt, ranging between 22 and 25 degrees at periods close to 41,000 years; (ii) changes in eccentricity of the Earth's orbit from a nearly circular to more elliptical form, which moderates the amplitude of the precession cycles, occurring at periods of 100,000 and 400,000 years; and (iii) precession or the change in season of closest approach to the sun of 19,000 and 23,000 year cycles (Pisias and Imbrie, 1986/1987; Komar, 1989; Solomon et al., 2007). These long-term variations in the Earth's orbit are the fundamental causes of ice-age-time-scale climatic cycles. Changes in the carbon budget and concomitant changes in atmospheric concentrations of carbon dioxide and other greenhouse gasses, which affect the Earth's retention of radiation, respond to Milankovitch variations and are part of the complex mechanism that brings the Earth in and out of ice ages (Bretherton, 1986/1987; Pisias and Imbrie, 1986/7).

There have been 10 major and 40 minor climatic oscillations over the last million years (Pisias and Imbrie, 1986/7). The most recent glacial period started about 116 ka (thousands of years ago), in response to orbital forcing, with the growth of ice sheets and fall of sea level culminating in the Last Glacial Maximum, at around 21 ka (between 30 and 19 ka) (Fig. 1.4) (Church

et al., 2001; Woodroffe and Horton, 2005; Jansen et al., 2007). The Holocene, the latest interglacial period, extends about 11.6 ka to the present (Jansen et al., 2007). Available evidence indicates that the Earth will not naturally enter another ice age for at least 30,000 years, and that current warming will not be mitigated by a natural cooling trend towards glacial conditions (Berger and Loutre, 2002; EPICA Community Members, 2004; Solomon et al., 2007).

Variations in global climate changes the amount of water stored in ice sheets. The largest factors affecting global sea-level over ice-age time scales are those related to changes in ice sheets through glacial-interglacial cycles, including vertical land movement from changes in the amount of water and ice on land and concomitant redistribution of mass within the Earth and oceans (Komar, 1989; Church et al., 2001; Woodroffe and Horton, 2005). Fig. 1.4 shows change in global sea-level over the past 140,000 years and contributions to this change from the volume of ice stored in the major ice sheets. Other factors affecting global sea-level over longer geological time scales include outgassing of the planet (over billions of years), and changes to the shapes of ocean basins during cycles of sea-floor spreading and plate tectonics (over tens to hundreds of millions of years) (Lambeck, 2004).

Eustatic sea level was about 4-6 m higher than present during the Last Interglacial period (about 123 ka, Fig. 1.4), based on measurements from direct sea level measurements using coastal sedimentary deposits and tropical coral sequences from tectonically stable locations (e.g. Rostami et al., 2000; Muhs et al., 2002). Based on results from oxygen-isotope analyses from the Bahamas (Chen et al., 1991), Seychelles (Israelson and Wohlfarth, 1999), Hawaii (Ku et al., 1990) and Western Australia (Stirling et al., 1995, 1998), sea-level reached a highstand of 2-10 m above present during the Last Interglacial. Observations of coral having been distributed further south than present along Australia's east and west coasts suggest that there were warmer temperatures during the Last Interglacial (Woodroffe, 2002).

During the past 140,000 years, during periods of rapid ice sheet decay, sea-level rise rates reached a high of  $40 \text{ mm a}^{-1}$  (Church et al., 2001). Since the last Glacial Maximum about 21 ka, sea level has risen about 130 m in regions distant from the major glaciation centers ('far-field' locations), as a result of loss of mass from ice sheets (an estimated  $50 \times 10^6 \text{ km}^3$  of ice melted



from land-based ice sheets) (Fig.1. 4) (Shackleton, 2000; Waelbroeck et al., 2002; Woodroffe, 2002; Siddal et al., 2003; Woodroffe and Horton, 2005; Peltier and Fairbanks, 2006). Peltier (2002) found sea-level to have been about 120 m below its present level at the Last Glacial maximum, based on coral-based records from Barbados. Woodroffe (2002) identified sea-level lowstand at a range of 103-175 m during the Last Glacial Maximum, based on a review of 11 globally distributed dated indicators. Waelbroeck et al. (2002) conducted a sea level reconstruction based upon coral records and deep sea O isotopes, corrected for abyssal ocean temperature changes, for the entire glacial-interglacial cycle, finding that the Last Glacial Maximum ice-equivalent eustatic sea level was about 120 m lower than present. The ice-equivalent eustatic sea level curve produced by Lambeck and Chappell (2001) shows a lowstand of about 140 m below present sea level. The maximum long-term sea-level rise rate since the Last Glacial Maximum was  $10 \text{ mm a}^{-1}$  (Church et al., 2001). Relative sea-levels fall by many hundreds of meters in regions that were once covered by the major ice sheets ('near and intermediate-field' locations) due to the isostatic rebound of the solid Earth following removal of the land-based ice (Woodroffe and Horton, 2005). Major melting of the ice sheets ceased about 6,000 years before present, and since then global sea-level has risen at a relatively slow rate of  $1\text{-}2 \text{ mm a}^{-1}$  (Fig.1. 4) (SCOR Working Group, 1991).

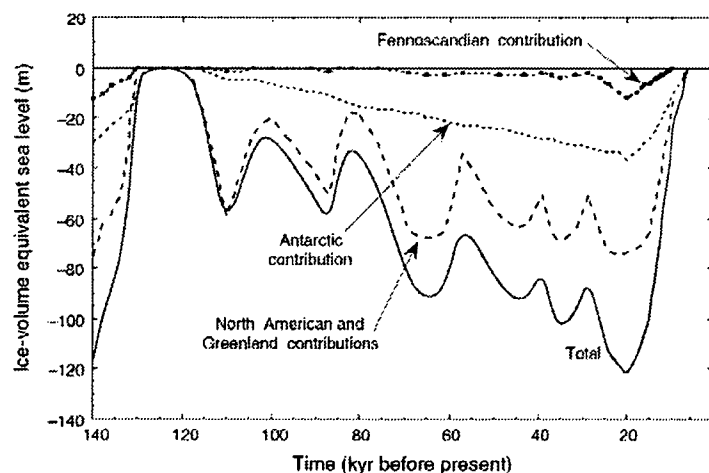


Fig.1.4. Estimated global sea-level change from 140,000 years before present and contributions from the Earth's major ice sheets (Church et al., 2001).

### **1.4.3. Holocene Sea-Level Patterns and Causes in the Pacific Islands Region**

Sea-level variability over Pleistocene time scales, resulting primarily from Milankovitch astronomical cycles, and resulting glacial cycles, is now well established (Section 1.4.2) and the general pattern of Holocene changes in sea-level in the Pacific is well understood. There is general agreement between modeled regional relative sea-level changes and field-interpreted changes on millennial time scales, since 6000 years BP, at most sites (Nunn and Peltier, 2001; Peltier, 2001; Goodwin, 2003).

In the Pacific Islands region, the dominant pattern of change in Holocene relative sea-level, at locations where local tectonism or lithospheric flexure are not predominant, was a uniform early Holocene rise in eustatic sea level, reaching its present level at 7000-6000 BP (years before A.D. 1950) in most areas, and a middle to late Holocene highstand in relative sea-level when sea-level was 1-2 m higher than present around 4000-2000 years BP (Grossman et al., 1998; Nunn and Peltier, 2001; Nunn, 2007). This middle to late Holocene high stand in relative sea-level occurrence in the Pacific Islands region is based on archaeological data and paleoshoreline records, as well as geophysical models of regional relative Holocene sea levels (Clark, 1990; Grossman et al., 1998; Dickinson, 2001). The high stand was caused by regional variation in the geoid and crustal motion (Grossman et al., 1998).

The highstand was followed by a regionally variable late Holocene fall in sea level relative to current sea-level, due to hydro-isostatic adjustment or equatorial ocean siphoning, and sea-level reached its present level between 4000-1200 BP (Grossman et al., 1998; Nunn, 1998a; Dickinson, 2001; Woodroffe, 2002; Goodwin and Grossman, 2003; Nunn, 2007). There is limited evidence that a second, smaller-magnitude highstand might have occurred in the late Holocene in some areas (e.g., Fiji, Nunn and Peltier, 2001). In general, small oceanic atolls experienced continued, smooth sea-level rise through the late Holocene, where the islands subside with the ocean floor (Hopley, 1987; Woodroffe and Horton, 2005). This is in contrast to larger land masses, which record a fall in relative sea-level through the late Holocene as sublithospheric flow beneath the sea floor is drawn to these land masses (Grossman et al., 1998; Woodroffe and Horton, 2005).

At about 700-550 BP there was a relatively fast rate of sea-level lowering, at about  $10 \text{ mm a}^{-1}$ , when sea-level fell a total of 50-80 cm, believed to have been caused by a drop in temperature, and referred to as the 'AD 1300 Event' (Nunn, 2007). The rate of change of Pacific Ocean mean (absolute) sea-level, based on 21 tide gauge records with  $\geq 24$  year length records in the 1950-2001 period, is estimated to be  $1.5 \text{ mm a}^{-1}$ , after making GIA/IB (glacial isostatic adjustment and atmosphere pressure – inverse barometer) corrections, consistent with the global average rate (Section 1.4) (Church et al., 2006; Bindoff et al., 2007; Solomon et al., 2007).

Hydro and glacio isostasy, equatorial ocean siphoning and lithospheric flexure are mechanisms that may have caused the relative sea-level positions in the Pacific (Goodwin, 2003; Grossman et al., 1998). Holocene sea-level records from the Pacific indicate evidence for a migrating geoid, through evidence of greater emergence in the central ocean basin interior during the mid to late Holocene, and recent relative sea-level fall in the central equatorial Pacific (Mitrovica and Peltier, 1991; Nunn and Peltier, 2001; Woodroffe and Horton, 2005). Sea-level observations also indicate that oceanic atolls less than 10 km in diameter were affected by hydro isostasy (Grossman et al., 1998). A geoidal process, called 'equatorial ocean siphoning', contributed to a fall in relative sea-level on Pacific Island coastlines in the late Holocene, in the last 3000 years (Mitrovica and Peltier, 1991; Woodroffe and Horton, 2005). During the Last Glacial Maximum, the weight of continental ice sheets caused downward deformation of the crust, forcing sublithospheric flow away. [A low latitude gravitational anomaly developed, creating a high in the oceanic geoid. Then, during the last glacial deglaciation, the continents viscoelastically rebounded, the gravity anomaly decayed and the oceanic geoid migrated from lower to higher latitudes].

Despite being distant from the major areas of glacial development and unloading in the Northern Hemisphere, GIA from deglaciation of Antarctica contributed substantially to changes in Holocene regional eustatic sea levels in Australasia and the southern Pacific region (Chappell, 1974; Hopley, 1987). Deglaciation in South Island of New Zealand and in Patagonia-Tierra del Fuego in southern Chile and Argentina may also have been large enough to cause an isostatic response (Hopley, 1987). The Antarctic ice sheet expanded to about  $37 \text{ million km}^3$  at the 18,000 BP Glacial Maximum, which

is about 9.8 million km<sup>3</sup> greater than the present area (Hughes et al., 1981; Hopley, 1987). This would have contributed about 25 m of rise in eustatic sea-level and likely caused substantial changes in global sea-level through GIA (Clark and Lingle, 1979; Hopley, 1987). For instance, substantial crustal rebound is indicated by a 23 m emerged shoreline at Wilkes Station, dated to 6,040 BP (Cameron and Goldthwaite, 1961). There are numerous tectonically active regions in the southern Pacific, which preclude use of reconstructions of their Holocene shorelines to provide information on GIA contributions to regional eustatic sea-level changes. However, there are some areas where insignificant vertical movement occurred in the Holocene, mainly at plate margins, for instance at Tongatapu and Eua in the Tonga group (Hopley, 1987).

While the general pattern of Holocene changes in sea-level in the Pacific is well understood, substantial gaps in knowledge remain regarding Pacific Holocene sea-level changes over centennial time scales (Nunn, 2007). On centennial time scales, the field-interpreted changes in relative sea-level show variability from the millennial linear trend (Goodwin, 2003). Based on records of Holocene sea-level change from different parts of the Pacific Islands region, the late Holocene sea-level fall followed different patterns, with different rates of lowering in different parts of the region (Dickinson, 2001). Because there are large site-specific deviations from the generalized regional model, detailed local studies are needed to understand site-specific Holocene sea-level changes (Dickinson and Green, 1998; Grossman et al., 1998; Goodwin and Grossman, 2003; Woodroffe and Horton, 2005).

Centennial-scale regional variability (non-linear regional changes in relative sea-level) is explained by tectonic processes, small eustatic contributions from variable pulses in global ice volume, and relatively small climate fluctuations and concomitant effects on ocean temperature and salinity (which affect steric sea-level) (Goodwin, 2003). For instance, short-term regional sea-level variability occurs in the Pacific from variation in wind strength and oceanic circulation associated with El Nino Southern Oscillation phases (Goodwin, 2003). Site-specific variability is caused by changes in sediment budgets, changes to wind, currents and wave energy and direction, and tectonic processes (in some cases, there are discrepancies of some areas having experienced coastal subsidence, while most areas in the region

experienced tectonic uplift), which cause site-specific differences in the timing and magnitude of the middle to late Holocene high stand and nature of sea-level fall (Clark, 1990; Dickinson and Green, 1998; Goodwin and Grossman, 2003; Woodroffe and Horton, 2005).

Local-scale tectonism and lithospheric flexure were mechanisms with substantial influence on relative Holocene sea-level changes in some locations in the Pacific. The tectonic variability in the Pacific Islands is evident from the height of the late Quaternary Last Interglacial terrace (Stage 5): far below current sea level on the Big Island of Hawaii, where isostatic submergence is occurring; at shallow depths on atolls, which are undergoing gradual subsidence; and above current sea level on Maketea Islands, which are undergoing lithospheric flexure (Ota and Kaizuka, 1991; Woodroffe, 2002). In Hawaii and Tahiti, horizontal movement of the Pacific Plate carries these islands across active volcanic hotspots (Pirazzoli and Montaggioni, 1988; Muhs and Szabo, 1994). Lithospheric loading at the hotspot causes a zone of subsidence under the area with volcanic accumulation, and an arch from flexure of the lithosphere causes uplifting several hundreds of kilometers from the zone of lithosphere downflexure (Dickinson and Green, 1998; Grossman et al., 1998). Rates of uplift due to lithospheric flexure from hotspot loading in Hawaii and the Society Islands have been estimated to be on the order of hundredths of a millimeter per year (Pirazzoli and Montaggioni, 1988; Muhs and Szabo, 1994).

Conflicting observations of past sea-levels at an individual site may result from employment of indirect evidence from shoreline indicators, which can provide unreliable age determinations as well as unreliable interpretations of past sea-levels, and employment of inconsistent methods preventing meaningful comparisons of results between studies (e.g., use of different water reference levels, failure to calibrate radiocarbon dates) (Ellison, 1989; Woodroffe and Horton, 2005). Some sea level indicators are definitive, while others are merely directional, and different indicators have different degrees of reliability in determining past sea-levels and dating (Ellison, 1989; Woodroffe and Horton, 2005). Paleoshoreline records of relative Holocene sea levels include observations of paleoshoreline notches in limestone seacliffs, which record paleo-high tide levels, emergent paleo-reef flats, which record paleo-low-tide levels, and emergent paleo-beachrock,

which records paleo-intertidal levels (Clark, 1990; Dickinson and Green, 1998; Dickinson, 2001; Goodwin and Grossman, 2003). These features do not provide definitive indicators of past sea-levels, due to their formation at a range of heights relative to sea-level, potential contamination, and difficulties in dating (Ellison, 1989; Nunn, 1998a). Use of debris such as intertidal shells as a paleo sea-level indicator also introduces potential error due to potential contamination (Kidson, 1982; Ellison, 1989; Nunn and Peltier, 2001). There have been numerous studies on Pacific atolls using coral species or coral microatolls as sea-level indicators (e.g., Bard et al., 1996; Grossman et al., 1998; Nunn and Peltier, 2001). Coral reefs are widely used to date sea-levels by radiocarbon or uranium series dating methods, where certain coral species' growth position are constrained by water depth (e.g., *Acropora* lives within 5 m of the surface), but the lower depth limit is poorly constrained (Kidson, 1982; Ellison, 1989; Woodroffe and Horton, 2005). Microatolls, however, have an indicative range as low as 3 cm, providing an extremely precise indicator of past sea-levels (Bard et al., 1996; Grossman et al., 1998; Woodroffe and Horton, 2005).

In summary, this review has highlighted that the general pattern of Holocene changes in sea-level in the Pacific is well understood, substantial gaps in knowledge remain regarding changes over centennial time scales, and site-specific Holocene sea-level changes can deviate substantially from generalized regional models. Due, in part, to the employment of inconsistent methods, and reliance on sea-level indicators with poor reliability, there is disagreement over short-temporal scale patterns in regional and site-specific Holocene sea-level. The next section presents the state of knowledge of Holocene sea-level changes in the Samoa archipelago, and specifically from Tutuila Island, American Samoa where the study sites investigated here are located.

#### **1.4.4. Sea-Level Research in the Samoa Archipelago**

Large site-specific deviations from generalized regional models of Holocene changes in relative sea level have been shown in many Pacific Islands. This includes the Samoan Islands (the independent country of Samoa, formerly Western Samoa) the largest islands being Savai'i and Upolu, constituting the Western portion of the Samoan archipelago (Dickinson and Green, 1998;

Grossman et al., 1998; Goodwin and Grossman, 2003; Woodroffe and Horton, 2005). In the Samoan Islands, research on Upolu and Savaii Islands supports there having been island emergence (Kear and Wood, 1959; Matsushima et al., 1984; Sugimura et al., 1988; Nunn, 1991, 1998a,b) and subsidence (Bloom, 1980; Dickinson and Green, 1998; Goodwin and Grossman, 2003) along different sections of coastline of these two islands since the middle Holocene. Goodwin and Grossman (2003) hypothesized that these conflicting observations are explained by there being complex and multiple local influences on coastal evolution, which are more significant than regional sea-level change. Reliance upon geomorphological features as sea-level indicators, which have wide indicative ranges and hence provide an imprecise relationship to former sea-level and can provide unreliable age determinations (Ellison, 1989; Woodroffe and Horton, 2005) may also partly explain these conflicting Holocene sea-level reconstructions. The possibility that inconsistent methods were employed by researchers, including the use of different water reference levels and calibration of radiocarbon dates (Woodroffe and Horton, 2005) is likely another factor contributing to the conflicting reconstructions.

Two studies in the Samoan Islands interpreted paleoshoreline indicators to document relatively high rates of submergence. Bloom (1980) interpreted submerged mangrove peaty mud from the south coast of Upolu Island to indicate that a high rate of local submergence occurred since the middle Holocene. However, while mangrove sediment only forms between mean sea-level and mean high water, the possibility that changes in elevation of the former mangrove sediment surface have occurred below the horizon from which the carbon dated sample was collected, such as through autocompaction of mangrove peat (Section 1.5.4), creates uncertainty in using the former elevation of the mangrove sediment surface to identify paleo-sea-level (Nunn, 1998a). In other words, it is possible that a proportion of the depth below current mean sea-level that the dated mangrove peat horizons were found to be located could have resulted from compaction of mangrove sediment below the sample depth, and not have entirely resulted from island subsidence as interpreted by Bloom (1980). If Bloom (1980) took the mangrove peat samples from the base of the paleo-mangrove sediment profile, and consolidated substrate was below this horizon, then

autocompaction would not have been a factor, and the basal peat would be a precise paleo-sea-level indicator. Dickinson and Green (1998) investigated an archaeological site containing prehistoric potsherds on Upolu, Samoa, located 2.25 m below modern sea level, dated to ca. 2750 BP. The sherd-bearing horizon was determined to be from Lapita pottery, made by the first inhabitants of the Pacific islands from the Bismarck Archipelago on the northwest to Tonga and Samoa on the southeast during 1500 – 500 B.C. (Kirch, 1997; Dickinson and Green, 1998). After accounting for GIA, estimated to be between 0.8 – 1.6 m relative to modern sea level at 2.8 ka (Mitrovica and Peltier, 1991), it was determined that the former coastline had subsided by about 4 m (range of 2.6 – 4.4 m) at a mean rate of continuous island subsidence of  $1.4 \text{ mm a}^{-1}$ , throughout the middle to late Holocene. Subsidence was hypothesized to have resulted from lithosphere downflexure from volcano loading on the island of Savai'i at the center of historic volcanism, due to Upolu Island's proximity to the seismically active Tonga Trench (Dickinson and Green, 1998; Dickinson, 2001). The use of Lapita pottery sherds as an indicator of a former shoreline has been disputed based on the argument that the Lapita homes may have been built over water and not on the beach, or the sherds may have been reworked from their original site of deposition (Nunn, 1991, 1998a).

Goodwin and Grossman (2003) identified a middle Holocene beachrock located within the modern intertidal range at their study site on the southern coastline of Upolu Island and interpreted its' position to constrain the relative sea-level history, at least for this section of the south coast of Upolu Island, to describe between 1-2.5 m of island subsidence since the middle Holocene. While this study also employs an indirect indicator of past relative sea-levels, the observation helps to further constrain the Holocene sea-level history of Upolu Island. These studies are consistent in interpreting paleoshoreline indicators to document that subsidence has occurred, but at different rates, along different portions of Upolu Island. Goodwin and Grossman's (2003) conflict with findings by Bloom (1980) and Dickinson and Green (1998), who suggested continuous island subsidence at a much higher rate during the same period for other areas of coastal Upolu Island, as previously explained, is possibly due to archaeological artifacts and mangrove peat being unequivocal indicators (Nunn, 1998a).



On Tutuila Island, American Samoa, the location of the mangrove sites included in this study, Daly (1924) was the first to report Holocene coastal evolution and relative sea-level history, in part, through observations of emerged benches and shore platforms, including a conspicuous emerged in situ reef in Vatia Bay, which reaches 3.2 m above present reef elevation. Stearns (1944) also identified emerged reef fragments on Tutuila, but neither he nor Daly (1924) identified the features' late Holocene dates. Nunn (1998a) hypothesized that eustatic sea-level changes over the past millennium as experienced on Tutuila, based on a comparison of elevated beach rock geomochronology with data from 18 other sites from the Pacific presumed to be relatively tectonically stable to include: (i) sea-level rise from 1050 – 690 BP, during a period of warming referred to as the Little Climatic Optimum; (ii) sea-level lowering from 575 – 150 BP, during the Little Ice Age; and (iii) rise over the past 150 years, a warming period. Nunn (1998a) reported six observations from two locations on Tutuila (Table 1.1), where the six paleoshoreline indicators of emerged in situ reef were between 0.75 m and 2.11 m above current reef level, dated to be between  $740 \pm 60$  and  $270 \pm 60$  BP. The four most recent paleoshoreline displacements from Tutuila, with dates ranging from 490 to 270 years BP, show about 2 m of emergence above that shown from New Zealand and Guam (Nunn, 1998a), again, documenting the site-specificity of Holocene relative sea-level changes. Nunn (1998a) hypothesizes that there might have been post-emergence contamination of the Tutuila paleoshoreline features, causing dates obtained for these features to be too young. Furthermore, as explained in the previous section, emergent paleo-reef do not provide definitive indicators of past sea-levels, due to their formation at a range of heights relative to sea-level resulting from a poorly constrained lower depth limit and the potential that contamination occurred (Ellison, 1989; Nunn, 1998a; Woodroffe and Horton, 2005). Comparing the elevations of the emerged reefs to the elevation of 'current reef level' (Nunn, 1998a) does not enable relating the elevations to past or current mean sea-level. Because reef elevation relative to mean sea-level may be different now than at the time the paleoshoreline reefs existed (e.g., due to the poorly constrained lower depth limit), and because it is possible that the tidal range has changed over the time period, the difference

in elevation between these features does not provide a definitive measure of the change in relative sea-level.

Table 1.1. Height of emergence above current reef level and age of six in situ reef from Tutuila, American Samoa (Nunn, 1998a).

Location on Tutuila, American Samoa	Emergence (m)	Age (BP)
Poloa	0.75	740 ± 60
Poloa	0.89	690 ± 70
Vatia Bay	2.11	490 ± 60
Vatia Bay	2.11	380 ± 60
Vatia Bay	2.11	350 ± 50
Vatia Bay	1.24	270 ± 60

Dickinson and Green (1998) observed 1.8 (± 0.1) m of shoreline emergence based on the differential elevation between a modern and emergent wave cut coastal bench at Ma’ama’a Cove, Aunu’u Island (located offshore of Tutuila Island). A date for the paleoshoreline feature was not reported. Again, a paleoshoreline wave cut bench is an unreliable indicator of past sea-level and date (Ellison, 1989; Woodroffe and Horton, 2005).

Results from analyses of mangrove stratigraphy offer some insights of Holocene relative sea-level changes. Based on radiocarbon dating of mangrove stratigraphy from cores taken from the seaward margins of five mangroves on Tutuila, the mangroves have maintained their approximate current seaward positions since between 270 – 3080 years BP when average sediment accretion rates for the five mangroves were between 1.2 – 4.5 mm a<sup>-1</sup> (Ellison, 2001). One site (Alofau) had a sediment accretion rate of 4.5 mm a<sup>-1</sup> since 270 BP (Ellison, 2001). A similar situation is evident at Masefau mangrove, which had a sediment accretion rate of 3.5 mm a<sup>-1</sup> since 410 BP (Ellison, 2001). The remaining three mangrove sites were observed to have accretion rates ranging between 1.2 – 2.0 mm a<sup>-1</sup> since between 620-3080 BP (Ellison, 2001). The observation of unchanged seaward margin position supports the inference that the mangrove sites kept pace with changes in relative sea-level. However, the likelihood that changes in elevation of the mangroves’ sediment surfaces occurred from a combination of several surface and subsurface processes (Section 1.5.4), and the relatively recent understanding that sedimentation rates are an imprecise method for determining trends in sea-level relative to a mangrove sediment

surface (Section 1.5.2.6) suggest that Ellison's (2001) observed accretion rates are potentially imprecise indicators of the American Samoa relative sea-level rise rate over the time periods. For instance, if autocompaction of mangrove sediment occurred below the depth of the cores, then the observed amount of accretion would underestimate the amount rise in relative sea-level.

Consistent observations of Holocene coastal emergence on Tutuila Island (Daly, 1924; Dickinson and Green, 1998; Nunn, 1998a) is explained by the location of Tutuila at a location where uparching of lithosphere between downflexures beneath more active volcanic islands to the east and west, where the principal volcanism occurred more than a million years ago (Clark and Michlovic, 1996; Dickinson and Green, 1998; Dickinson, 2001), and the occurrence of the Little Ice Age during the period of the dates of the paleoshoreline features, when at about 575 BP regional relative sea-level rose a bit above present sea-level and then gradually fell to about -0.9 m below present sea-level (Nunn, 1998a).

In the Manua'a Islands group of American Samoa (comprised of Ofu, Olosega, and Ta'u islands, the eastern portion of American Samoa), Stice and McCoy (1968) found coastal features documenting approximately 2 m of shoreline emergence in the Manua'a Islands, but dates of these features were not obtained. Dickinson and Green (1998) observed 2.0 m (1.2 – 2.8 m) of submergence on Ofu Island, based on the position of mid-Holocene paleoshoreline indicators. Subsidence observed from archaeological sites in the Manua'a Islands of American Samoa is hypothesized to be caused by there being recent volcanism to cause subsidence by volcanic loading (Kirch, 1993a,b; Dickinson and Green, 1998; Dickinson, 2001). Again, none of these paleoshoreline indicators are definitive.

In the Pacific, there is limited definitive documentation of mid to late Holocene eustatic sea-level changes (Nunn, 1998a; Woodroffe and Horton, 2005). As in other parts of the Pacific, available evidence indicates that Holocene sea-level changes in the Samoa archipelago over centennial time scales have been highly site-specific, with different islands, and in some cases, different sections of coastline along individual islands, exhibiting substantially different patterns. Reliance on indicators of Holocene sea-levels that provide imprecise relationships to former sea-level, potential error

in dating, and possible inconsistency in methods, likely explains some of the conflicting Holocene sea-level reconstructions in the Samoa archipelago. Based on available evidence, it is likely that indicators dating a Holocene sea-level maximum of about 1.5 m above present from some locations in the Samoan Islands (e.g., emerged beachrock located between 0.8 – 0.95 m above current mean sea level on Savai'i and Upolu Islands, a conglomerate-filled wave-cut notch in basalt at 2.3 m above current mean sea level on Upolu Island, and dating of coral sand and shell fragments from about 1-2 m above current mean sea level on Sava'i and Upolu Islands [Grant-Taylor and Rafter, 1962; Roda, 1988; Sugimura et al., 1988]) and parts of American Samoa (Daly, 1924; Stice and McCoy, 1968; Kirch et al., 1990; Nunn, 1998a) are accurate (Grossman et al., 1998). These observations are generally consistent with model predictions for Holocene sea-level changes in the Samoa Archipelago, predicting that postglacial isostatic adjustments would have resulted in a mid-Holocene relative sea-level highstand of between 1.9-2.6 m at 4000 BP (Fig. 8 in Mitrovica and Peltier, 1991).

The understanding of mangrove vulnerability and predicted responses to relative sea-level rise over human time scales benefits to a degree from understanding of Holocene relative sea-level changes in American Samoa, and long-term (centennial to millennial) trends shown by paleoenvironmental reconstructions of mangroves to past sea-level fluctuations. However, as will be discussed in the following section, contemporary information on relative sea-level trends, in combination with information on three additional factors, is needed to accurately assess mangrove vulnerability over recent decades and predict responses over coming decades (Sections 1.5.2.3 and 1.5.2.5). The Permanent Service for Mean Sea Level and the University of Hawaii Sea Level Center Joint Archive for Sea-Level and GLOSS/CLIVAR Research Quality Data Set databases provide data, corrected for changes in local datum, from the continuously monitored Pago Pago tide gauge on Tutuila Island, American Samoa, beginning in October 1948. Recent estimates of sea-level change from the 21<sup>st</sup> Century have been estimated using these tide gauge observations. Church et al. (2006) determined a relative sea-level rise rate of  $1.6 \pm 0.3 \text{ mm a}^{-1}$  for Pago Pago, American Samoa, based on analysis of tide gauge data from 1950 – 2000, which translates to  $2.1 \text{ mm a}^{-1}$  after correcting for glacial isostatic adjustment and atmospheric pressure effects

(the 'inverse barometer' response of the ocean to variations in atmospheric pressure). The individual components contributing to the observed rate of relative sea-level rise measured by the Pago Pago tide gauge are described in Section 5.1. Section 1.5.2.6 explains the need for observing trends in sediment surface elevation of mangroves in the context of understanding the threat to mangroves from sea-level rise, and Section 1.5.6 describes alternative approaches to measure trends in sea-level relative to the mangrove sediment surface.

## **1.5. MANGROVE VULNERABILITY AND RESPONSES TO RELATIVE SEA-LEVEL RISE**

### **1.5.1. Mangrove Geomorphic Settings**

The classical, successional model of mangrove dynamics, where mangroves move towards a climax community through sediment accumulation, and the plant community transitions from a pioneer to a climax community, has been replaced with a model that accounts for the diversity in landform classes, site-specific processes, complex environmental settings and geomorphic evolution (Davis, 1940; Chapman, 1944; Egler, 1952; Thom, 1984; Ellison and Stoddart, 1991; French, 1991; Duke et al., 1998; Ellison, 1993, 2000, 2001; Woodroffe, 1995, 2002; Alleng, 1998; Lucas et al., 2002).

Observations of vegetation zonation in mangroves was initially believed to support the existence of succession, however, zonation or mosaics of mangrove communities, is alternatively understood to represent each species' optimal niche for productivity (Egler, 1952; Woodroffe, 1992, 2002).

This alternative model views mangroves as opportunistic, colonizing available substrate in an ecological response to various external conditions, where mangrove species distribution is determined by individual mangrove species having specific tolerance levels to environmental factors, including hydrologic and salinity regimes; wave energy; soil and water pH; sediment composition and stability; nutrient concentrations; and degree of faunal predation (Egler, 1952; Thom, 1984; Woodroffe, 1992, 2002). This species-specific tolerance to various environmental parameters results in mangrove species zonal distribution and determines if an individual mangrove species

can establish and survive in a specific location (Tomlinson, 1986; Naidoo, 1985, 1990; Duke, 1992; Wakushima et al., 1994a, 1994b; Duke et al., 1998).

This understanding of mangroves as opportunistic colonizers with distribution controlled through ecological responses to environmental factors highlights the importance of the geomorphic setting in determining where mangrove ecosystems establish, their structure and functional processes. An understanding of a mangrove's geomorphic setting, including sedimentation processes (sediment supply and type), hydrology, and energy regime, is likewise important in understanding responses to changes in site specific relative sea-level, as these affect both surface and subsurface controls on mangrove sediment elevation. Mangrove forests occupy an inter-tidal habitat, and are extensively developed on accretionary shorelines, where sedimentation and flooding/drainage occurs through a network of tidal channels, and where surface and subsurface processes determines their ability to keep up with sea-level rise (Woodroffe, 2002). Mangroves are found in a range of physiographic settings where there is adequate protection from wave action, including protected shallow bays, protected estuaries, lagoons, leeward sides of peninsulas and islands, and behind spits, coastal and barrier dunes, and offshore islands (Chapman, 1976; Mitsch and Gosselink, 1993). Surface elevation change may be influenced by variables such as sediment type, sediment supply, hydrology and energy regime (tides, floods, wave energy) (Section 1.5.4) (Cahoon and Hensel, 2006).

These variables are closely associated with a site's coastal geomorphology. Mangroves have been classified according to different hydrologic and topographic settings (Lugo and Snedaker, 1974; Wharton et al., 1976; Lugo, 1980), simplified by Cintron et al. (1985) and Mitsch and Gosselink (1993) to include four main categories:

- (i) **Fringe and Overwash:** Fringe mangroves are found on protected shorelines as well as along lagoons and rivers, and are exposed to daily tides. They tend to be a net sink for organic matter due to low-energy tides and presence of dense aerial roots, which dissipates tidal and wave energy, and due to their direct exposure to wave and wind energy during storms, which can bring debris into the ecosystem. Overwash mangrove islands occur on small islands and spits, and are completely overwashed at high tide. In overwash mangroves, tidal energy is high enough to

remove most loose debris and leaf litter and transport it to adjacent coastal ecosystems.

- (ii) **Riverine:** Riverine mangroves are tall mangrove stands that flank channels of coastal rivers and streams, and can extend some distance inland from the coast. They may lack inundation for long periods of time during the dry season or episodic periods of low precipitation. Due to high levels of freshwater and nutrient inputs, they have high primary productivity and export substantial amounts of organic matter. Salinity is on average lower than in other mangrove classes, as during periods of high rainfall during wet seasons, freshwater flushing leaches salt from sediments.
- (iii) **Basin:** Basin mangroves occur near the coastline in inland depressions, such as behind fringe mangroves, and in drainage depressions where water is stagnant or flowing slowly. The mangrove trees are often low in stature. The mangrove systems are typically exposed to tidal inundation only during extreme high tides. The prevalent conditions of stagnant standing water and low water turnover rate results in soils with high salinity and low redox potential.
- (iv) **Dwarf, Scrub and Hammock:** These are relatively low productivity systems occurring in nutrient-poor conditions, hypersaline soils, and/or at the higher latitudinal limits of mangrove distribution. These low-productive mangroves might lack nutrients or freshwater inputs, and are dominated by short trees at relatively low densities. The soil might be sandy or a limestone marl. They might be exposed to the tides only during extreme high tides or storm surges, and can be flooded from freshwater runoff during rainy seasons. At sites located at the geographical limits of mangrove distribution, dwarf mangroves might exist in riverine, fringe or basin settings. Hammock mangroves are outposts of mangroves within the Everglades (Florida, USA) environment, they occur as isolated and slightly elevated islands (higher elevation developed through long-term peat accumulation) in the coastal fringe, with periodic tidal exposure, overlapping conditions of basin and scrub mangrove classes.

Across these classes, the relative dominance of rivers and tides, the predominant sources of flooding, result in disparate system functioning (Woodroffe, 2002). In sites where river flow is the dominant flooding process, sediments are deposited from river water, while organic production is augmented by the freshwater input, and this organic matter is exported through river flow (Fleming et al., 1990; Furukawa et al., 1997; Woodroffe, 2002). At sites where tides dominate flooding processes, there is bi-directional flux, transporting organic matter from the mangrove to adjacent coastal areas, but to a more limited extent than in riverine settings (Jiminez and Sauter, 1991; Lee, 1995, 1999; Woodroffe, 2002). Finally, interior sites are generally remote from direct flushing, are sinks for organic matter, nutrients and sediment, and export little sediment or organic matter (Twilley, 1985; Woodroffe, 2002).

The greater the hydrologic turnover, the higher the primary productivity, where these four classes range from highest to lowest hydrologic turnover and productivity in the following order: riverine, fringe, basin, dwarf/scrub (Twilley, 1988). The higher the salinity, the more energy a mangrove community expends for maintenance in these conditions, through processes that exclude and eliminate salt and retain freshwater. Freshwater inputs from upland sources and tidal exchange, which affect water chemistry, have been demonstrated to be important factors controlling primary productivity and general mangrove function (Lugo and Snedaker, 1974; Lugo, 1990). In particular, tidal influences on the duration, frequency and depth of inundation exhibit a substantial influence on the extent and functioning of mangroves (Lewis, 2005).

An alternative to the ecologically-based classification scheme of Cintron et al. (1985), as modified from Lugo and Snedaker (1974), Wharton et al. (1976) and Lugo (1980), is a classification that focuses on differences in geomorphology, where each geomorphologically defined class possesses distinct sedimentological, geochemical and functional characteristics (Woodroffe, 2002). Thom (1982) described five geomorphic classes where mangroves occur:

- (i) **River dominated allochthonous:** These allochthonous (sediments are derived primarily from outside the wetland ecosystem) coasts of low tidal range exist where river discharge of freshwater and sediment leads to



rapid deposition of terrigenous sands, silts and clays to form deltas. The delta has multiple branching distributaries forming elongate protrusions, creating a coastline of shallow bays and lagoons between distributaries. This class has a high degree of morphologic diversity and rapid habitat change.

- (ii) **Tide-dominated allochthonous:** In this class, the dominant physical process is high tidal range and strong bidirectional tidal currents. The currents disperse sediment brought to the coast via rivers, and form linear elongate sand bodies in the offshore zone. Wave energy is typically low due to frictional attenuation over broad intertidal shoals.
- (iii) **Wave-dominated barrier-lagoon** (autochthonous, sediments are produced primarily within the wetland ecosystem): This class has relatively very high wave energy and relatively low amounts of river discharge. Offshore barrier islands, barrier spits or bay barriers are typical in this class. Barrier islands enclose broad elongate lagoons, while bay barriers enclosed drowned river valleys. Small deltas prograde into these water bodies.
- (iv) **Composite river and wave dominated:** This class has a combination of high wave energy and high river discharge. Sand discharged from the river is redistributed by waves alongshore to form extensive sand sheets. The coastal plain has sand beach ridges with narrow discontinuous lagoons.
- (v) **Drowned river valley:** This class is defined by a bedrock valley system which has been drowned by relative sea-level rise. Sediment accretion has not been sufficient from marine or river sources to infill the open estuarine system. At the mouth of the drowned valley bordering the open sea, a tidal delta may occur composed of marine sand, reworked landward during a marine transgression.

Subsequently, Thom (1984) identified an additional three carbonate classes, where terrestrial sediment supply is low or absent, and where calcareous sediment production dominates: carbonate platform, sand/shingle barrier, and Quaternary reef top. Mangroves of oceanic islands, coral reefs and carbonate banks typically are in carbonate settings with a small tidal range

and where the autochthonous mangrove sediment is made of peat (Woodroffe, 1992).

The geomorphic classification scheme developed by Woodroffe (2002), which incorporates numerous previous classifications (Thom, 1982, 1984; Dijkema, 1987; Allen and Pye, 1992; Woodroffe, 1992; French, 1997; Allen, 2000), includes the following geomorphic settings where mangroves and salt marshes occur:

- (i) **Back barrier:** Mangroves occur in back-barrier and lagoonal settings, where the mangroves receive some protection from direct wave energy.
- (ii) **Embayment:** Mangroves can occur in large, shallow coastal bedrock embayments, which can have a restricted entrance, and might receive large freshwater inputs. Embayment mangroves may overlap with the drowned valley category.
- (iii) **Estuarine:** Estuarine mangroves are extensive where there is a large tidal range, or where there is large clastic sedimentation that is reworked primarily by tidal action. Within an estuary, mangroves occur in several geomorphologic habitats.
- (iv) **Deltaic:** Deltaic mangroves occur in river-dominated settings in association with large rivers. As with estuarine mangroves, within a delta, mangroves occur in several geomorphologic habitats. Mangroves are extensive on tide-dominated deltaic plains.
- (v) **Open coast:** Mangroves can occur along open coasts, especially along the landward margin of mudflats and chenier ridges, which attenuate wave energy. However, mangroves tend not to be extensive on wave-dominated coasts. Wave action causes the sediment to be sandy.
- (vi) **Drowned valley.** As defined by Thom (1984), above, mangroves occur in drowned valleys, which are bedrock valley settings that have been drowned by relative sea-level rise.

In conclusion, an understanding of a mangrove's geomorphic setting, including the hydrology, topography, sources of flooding and the hydrologic turnover rate, and sedimentation processes, including sediment type and supply, aids in understanding past and predicting future responses to site-specific relative sea-level rise. This is the case because site-specific geomorphic parameters affect the factors controlling a mangrove's sediment

surface elevation (Section 1.5.4) as well as feedback mechanisms to changes in site-specific relative sea-level (Section 1.5.5).

However, there are two general limitations of employing geomorphic classifications to understand the threat to a mangrove site from sea-level rise. Geomorphic classifications for individual mangrove sites are based on a site's primary overall setting characteristics. But an individual site can contain numerous categories of settings nested within the site, which differ from the general overall site classification. Woodroffe (1987, 1992, 2002) recognized that these broad geomorphic categories are not mutually exclusive, such that an individual site might fall between two or more individual geomorphological settings, and also might occur in a combination of settings. These categories can be further split into more detailed subclasses. For instance, deltas can be sub-divided into those that are dominated by wave, river or tidal processes (Thom, 1984), while estuarine mangroves can be subdivided into those that are within embayments with restricted entrances, estuarine fringe and estuarine back-barrier (Woodroffe, 2002). These subclasses will have disparate sediment properties, and the distinctive geomorphic context will determine the specific habitat characteristics and vegetation structure. Furthermore, the microtopography of a particular landform, including river levees, beach-ridge swales, and chenier ridges, affect the local landward extent of the tidal range and plant establishment and growth (Thom, 1984; Semeniuk, 1994). As a result of there being multiple geomorphic settings within a site, there is the potential that sites that are lumped under one category will have substantially disparate degrees of vulnerability and responses to the same rates of relative sea-level rise.

A second limitation relates to the poor state of development of reliable predictive elevation models for mangrove ecosystems. Models used to predict salt marsh elevation responses to sea-level rise projections (Morris et al. 2002, Rybczyk and Cahoon 2002) have not been developed for mangrove systems. There likewise is a lack of reliable predictive erosion models for mangroves. Calculation of mangrove erosion using a predictive model of beach erosion called the Bruun rule (Bruun, 1962, 1988) is expected to produce inaccurate results because mangroves are not expected to respond in accordance with Bruun rule assumptions, and the Bruun rule, as with other

general predictive models of coastal erosion, is not suitable for small-scale, site-specific estimates (Section 5.4) (Bruun, 1988; List et al., 1997; Komar, 1998; Pilkey and Cooper, 2004). There has been a lack of an observed correlation between geomorphic class and change in mangrove sediment elevation from an ad hoc global database: Cahoon and Hensel (2006) found no significant correlations between geomorphic class and sediment elevation trends, based on a review of 28 mangrove study sites employing the surface elevation table (SET) method (Boumans and Day, 1993), from the Wider Caribbean and Western Pacific regions. Given this state of knowledge of relationship between geomorphic setting and change in sediment surface elevation, until reliable predictive elevation models are developed for mangrove ecosystems, site-specific monitoring is necessary to assess vulnerability to sea-level rise.

#### **1.5.2. Observations of Mangrove Vulnerability and Responses to Changes in Sea-Level**

This section reviews paleoenvironmental reconstructions, recent observations of changes in coastal wetland position, and observations of trends in sedimentation and sediment surface elevation in order to describe the state of knowledge of mangrove vulnerability and responses to changes in sea-level. Table 1.2 provides a range of examples from the literature of observations of changes in coastal wetland position, trends in sedimentation, or trends in sediment surface elevation, and changes in regional or site-specific relative sea-level.

In this section, we first employ the body of literature reviewed in Table 1.2 to explain how coastal wetlands located in regions experiencing relative sea-level rise can exhibit disparate responses in changes in position. In some cases coastal wetlands were observed to have not kept pace and migrate landward, explained by sea-level having risen relative to the elevation of the wetland's sediment surface (referred to in this thesis as 'site-specific' relative sea-level rise). Other sites were observed to have kept pace and maintained position, explained by there having been no change in site-specific relative sea-level. A third category of sites were observed to prograded seaward, explained by there having been a fall in site-specific sea-level. Then, predominant factors controlling mangrove position are

discussed. The threat to coral reefs from relative sea-level rise is reviewed to document parallels to mangrove ecosystems. Lessons learned and gaps, and limitations with previous research are described for paleo-shoreline reconstructions and observations of trends in sedimentation and sediment surface elevation. Finally, we summarize the state of knowledge and gaps in understanding of mangrove vulnerability and responses to changes in sea-level.

Table 1.2. Range of examples of responses of coastal wetlands located in regions experiencing relative sea-level rise, including observations of position, trends in sedimentation, or trends in sediment surface elevation.

Location	Setting	Method	Timescale	Results	Citation
<b>Rate of change in sediment elevation &lt; regional relative sea-level rise rate</b>					
<b>Site-specific relative sea-level rise, wetland did not keep pace with regional relative sea-level rise and in some cases migrated landward</b>					
Barataria Basin, Louisiana, U.S.A.	Continental, streamside and inland coastal marsh	$^{137}\text{Cs}$ activity dating	Decades	Located in area experiencing rapid subsidence, streamside and inland marshes had accretion rates of 13.5 and 7.5 mm a <sup>-1</sup> , respectively. The inland marsh was hypothesized to not be keeping pace with regional relative sea-level rise based on the observation of small areas of marsh being converted to open water.	DeLaune et al., 1978
Pataguanset River estuary, East Lyme, Connecticut, U.S.A.	Continental estuary	Radiocarbon dating of basal peat, rhizome species identification	Thousands of years	Brackish marsh replaced freshwater wetlands between 3800-4000 BP during a period of regional relative sea-level rise. Sediment accretion at a rate of 1.1 mm a <sup>-1</sup> was presumably insufficient to keep pace with the rate of sea-level rise. Gradually, by 3000 BP, salt marsh had replaced submerging brackish marshes, adjacent uplands, and accreting tidal flats.	Orson et al., 1987
Tongatapu Island, Tonga	Low island mangrove, carbonate setting	Radiocarbon dating of stratigraphy, pollen analysis	Thousands of years	A large mangrove persisted 7000-5500 BP during sea-level rise of 1.2 mm a <sup>-1</sup> . When the rate increased, the mangrove retreated and persisted as a narrow fringe at about 5500 BP, and later re-colonized the entire original area following lowering of relative sea-level in the late Holocene.	Ellison, 1989
Hungry Bay, Bermuda	Low island mangrove, carbonate	Radiocarbon dating of stratigraphy, $^{210}\text{Pb}$	Decades, thousands of years	The mangrove, which established in the last 3000 years when regional relative sea-level rise rate slowed from 2.6 to 0.7 mm a <sup>-1</sup> , began to retreat at 1000 BP when the	Ellison, 1993, 1996a

	setting	dating, pollen analysis, surveying		regional sea-level rise rate increased to 1.4 mm a <sup>-1</sup> , and lost 26% of its area over just the last century due to the retreat of its seaward edge when the regional relative sea-level rise rate was 2.8 mm a <sup>-1</sup> . Surveys demonstrated that the elevation of the sediment surface of the mangrove at its seaward margin was lower in the tidal spectrum (below MSL) than typical, confirming that the mangrove area at the seaward margin was under stress from site-specific relative sea-level rise.	
Nag Creek and Rumstick Cove, Nargansett Bay, Rhode Island, U.S.A.	Continental, salt marsh	Quadrat monitoring of vegetation zone boundary position, radiocarbon dating of stratigraphy, plant macrofossils, <sup>210</sup> Pb and <sup>137</sup> Cs dating	Years and thousands of years	Low-marsh vegetation was observed to migrate landward and displace higher-marsh species in two salt marshes over a four-year period, where accretion rates have been less than the regional relative sea-level rise rate of 2.7 mm a <sup>-1</sup> . The initiation of landward migration of the salt marsh vegetation zones was in the late 19 <sup>th</sup> century, when the regional rate of relative sea-level rise accelerated.	Donnelly and Bertness, 2001
Southern Irian Jaya, New Guinea	High island macrotidal estuarine mangrove	Radiocarbon dating of stratigraphy, pollen analysis	Thousands of years	A <i>Bruguiera</i> zone (typically found in mangrove landward margins) was observed to have been replaced by <i>Rhizophora</i> (typically found at mangrove seaward margins) about 3000 years ago, indicating that the mangrove gradually migrated landward during the late Holocene, when relative sea-level rose at a relatively slow rate of 0.7 mm a <sup>-1</sup> .	Ellison, 2005
Western Port Bay, Victoria, Australia	Continental estuarine mangroves and salt marshes	SET-MH (surface elevation table – marker horizon) technology, Excess <sup>210</sup> Pb	Years, decades	Salt marsh subsidence occurred due to reduced groundwater inputs associated with El Nino phases, contributing to the site-specific rise in relative sea-level, allowing mangroves to migrate landward into salt marshes. Salt marshes with relatively high rates of increase in	Rogers et al., 2005a

activity dating,  
aerial photography  
interpretation

sediment elevation were observed to have low rates of mangrove encroachment, while salt marshes observed to be lowering in elevation had high rates of mangrove encroachment. Sediment accretion rates significantly exceeded rates of surface elevation change in the mangroves. There was no change in site specific relative sea-level over three years at the four mangrove study sites.

North Sound, Grand Cayman	Low island mangrove, carbonate setting	Radiocarbon dating of stratigraphy	Thousands of years	Between 4080 and 3230 BP, a seaward mangrove margin migrated landward as regional relative sea-level rose 2.8-3.3 mm a <sup>-1</sup> .	Ellison, 2006
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**Rate of change in sediment elevation = regional relative sea-level rise rate**

**No change in site-specific relative sea-level, wetland maintains position while regional sea-level rises**

Farm River, New Haven, Connecticut, U.S.A.	Coastal salt marsh	Excess <sup>210</sup> Pb activity dating	Hundreds of years	A salt marsh kept pace with the rate of regional relative sea-level rise when the marsh experienced vertical accretion through peat development at a rate of 1.6 mm a <sup>-1</sup> over 203 years.	McCaffery and Thompson, 1980
South San Francisco Bay, California, U.S.A.	Coastal salt marshes	<sup>137</sup> Cs activity dating	Decades	Marsh sediment accretion and peat formation offset subsidence to enable the salt marsh to keep pace with a rate of regional relative sea-level rise of 1.3 mm a <sup>-1</sup> , allowing the marsh sediment elevation to persist at ≥ 0.1 m above mean high water.	Patrick and DeLaune, 1990
Rookery Bay, Southwestern Florida, U.S.A.	Continental mangroves (fringe, basin and overwash geomorphic	SET-MH technology	Years	Four mangrove sites were observed to experience a gain in sediment surface elevation over a 1– 2.5-year period at rates ranging between 1.4 – 3.7 mm a <sup>-1</sup> , enabling the mangroves to keep pace with regional relative sea-level rise	Cahoon and Lynch, 1997



settings)

Port Royal, Jamaica	Low island, microtidal fringing mangrove	Interpretation of historical maps and aerial photographs	Hundreds of years, decades	A mangrove site was observed to have been generally stable in area over 300 years, where the regional relative sea-level rise rate was $1 \text{ mm a}^{-1}$ .	Alleng, 1998
Amherst Point, Nova Scotia, Canada	Coastal salt marsh	Radiocarbon dating of stratigraphy, plant macrofossils, foraminiferal analysis, pollen analysis	Thousands of years	The landward edge of a coastal marsh was observed to have vertically accreted 7.5 m at a mean rate of $2.6 \text{ mm a}^{-1}$ since about 2850 BP, as regional relative sea level was rising at about $1 \text{ mm a}^{-1}$ . The increase in elevation of the wetland sediment surface was, on average, equal to or greater than the rise in local sea level, i.e. site-specific sea-level was falling or not changing. High water levels increased at varying rates, and the rate of salt marsh accretion lagged behind when increase was rapid, and caught up when water level rise was less rapid, laying down more organic rich horizons.	Shaw and Ceman, 1999
Meso- American Barrier Reef system, Caribbean: Twin Cays and Cat Cay, Belize; Bay Islands, Honduras, Isla San Cristobal, Panama	Mangrove islands	Radiocarbon dating of peat stratigraphy, SET- MH technology, controlled nutrient addition experiment	Years to thousands of years	(i) Seaward mangrove margins were observed over a three year period to have a trend in gain in elevation of the sediment surface of $4.1 \text{ mm a}^{-1}$ , while mangrove interior and landward margins were observed to lower in elevation at rates of $-1.1$ and $-3.7 \text{ mm a}^{-1}$ , respectively, which, when compared to estimates of site-specific relative sea-level rise, suggested that the seaward zone is likely keeping pace, while the landward zones are not.  (ii) Peat thickness from multiple sites ranged from 0.4 – 10.0 m, where peat accumulation closely followed rates of regional relative sea-level rise. At Twin Cays, Belize, mangroves did not exist when regional relative-sea level rise rate exceeded $5 \text{ mm a}^{-1}$ , but established when rates	McKee et al., 2007

were about  $3.5 \text{ mm a}^{-1}$ . Mangrove sites at Honduras and Panama established when regional relative sea-level was about  $1 \text{ mm a}^{-1}$ .

**Rate of change in sediment elevation > regional relative sea-level rise rate**

**Site-specific relative sea-level lowering, and in some cases wetland prograded seaward and possibly laterally**

Homebush Bay, Australia	Continental mangrove	SET-MH technology	Years	Reduced groundwater inputs associated with El Nino phases was inferred to cause reduced rates of rise in sediment elevation. Sediment elevation exceeded sediment vertical accretion during the study period. On average, the mangrove exhibited site-specific relative sea-level lowering during the 3.5 year period. Analysis of aerial photographs documented an increase in mangrove area of 14 ha, and reduction in salt marsh area by 61 ha, from 1930 – 2000. Information is not presented on trends in change in position of the mangrove seaward margin.	Rogers et al., 2005b
Richmond River Estuary, Australia	Continental microtidal, estuarine, mangrove	Radiocarbon dating of stratigraphy, pollen analysis	Thousands of years	From 7000 to 6000 BP an estuarine mangrove became established and persisted during a period of moderate, decelerating regional relative sea-level rise. Subsequently, after 6000 BP, when sea-level stabilized, the substrate elevation rose above intertidal elevations, and the mangrove was displaced by freshwater swamp and dryland vegetation communities. The rate of regional relative sea-level rise was estimated to have been $5 \text{ mm a}^{-1}$ from 7000 -- 6500 BP, and reached present sea-level sometime between 6500 – 6000 BP. Rhizophoraceae had greater dominance during this mid-Holocene transgressive period than present.	Hashimoto et al., 2006
Multiple trends in site-specific relative sea-level South Alligator River,	Continental, macrotidal	Radiocarbon dating of	Thousands of years	Extensive mangrove development was observed to occur in the mid-Holocene from 8000-6800 BP during a	Woodroffe et al.,

Northern Australia	estuarine mangrove	stratigraphy, pollen analysis		transgressive phase when the mangrove experienced site-specific relative sea-level rise, mangrove forest migrated landward into a valley as regional relative sea-level rose to reach its current position. As the rate of regional sea-level rise slowed, mangrove established over an 80,000 ha area, which existed from about 6800 – 5300 BP. Subsequently, sediment accretion continued, resulting in site-specific relative sea-level lowering, converting most of the mangrove habitat to grass and sedge-covered floodplains. The study does not document whether the mangrove have migrated seaward during this latter phase.	1985, 1989
West Alligator River, Australia	Continental, macrotidal estuarine mangrove	Aerial photograph interpretation	Decades	Over 41 years, as regional relative sea-level rose, mangroves on the western bank of the river mouth prograded seaward at a mean rate of $6 \text{ m a}^{-1}$ , and tide channels narrowed in width as mangroves colonized mud banks. During the same period, mangroves on the eastern bank of the same river retreated landward at a mean rate of $2.4 \text{ m a}^{-1}$ , and tide channels widened as mangroves retreated away from the channels.	Lucas et al., 2002
Abatan river, Southwest Bohol, Philippines	Continental microtidal mangrove	Radiocarbon dating of stratigraphy, interpretation of tidal notches, foraminiferal analysis	Thousands of years	From 8000 to 6000 BP regional relative sea-level rose rapidly from 12 to 2 m below present level. From 6000 to 4500 BP the rate of regional relative sea-level slowed, rising from 2 to 1 m below present sea-level, and mangroves became established. Then, from about 4500 to 4300 BP, the rate of rise of regional sea-level accelerated, rising rapidly to 0.7 m above present sea-level, and during this period of transgression mangroves were not able to keep pace. Regional sea-level then remained stable until about 2500 BP, at which point regional sea-level began to fall to the present level, when extensive mangrove peat formation occurred across a broad, shallow basin.	Berdin et al., 2003

Yela, Utwe and Enipoas swamps, Pohnpei and Kosrae, Federated States of Micronesia	High island fringe, riverine and interior mangroves	SET-MH technology	Years	Change in elevation of the sediment surface of mangrove vegetation zones over 2.5 years ranged from -0.2 to 3.4 mm a <sup>-1</sup> , with an average of 1.3 mm a <sup>-1</sup> , accounting for controls on elevation to a depth of 1 m. Based on analysis of tide gauge data from Pohnpei, relative sea-level has been rising at a rate of 1.8 mm a <sup>-1</sup> (Church et al., 2006), indicating that certain zones of the mangrove study sites were experiencing site-specific sea-level lowering (where the rate of change in elevation > 1.8 mm a <sup>-1</sup> ) while other zones of the same mangrove were experiencing site-specific relative sea-level rise. Study results document that mean sediment vertical accretion was greater than mean sediment elevation.	Krauss et al, 2003
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#### **1.5.2.1. Wetland Elevation Does Not Keep Pace with Rate of Rising Sea-Level**

Several studies have documented mangroves of both low islands and continental margins not having kept pace with relative sea-level rise (e.g., DeLaune et al., 1978; Orson et al., 1987; Grindrod et al., 1999; Woodroffe et al., 1985, 1989; Hanebuth et al., 2000; Donnelly and Bertness, 2001; Lucas et al., 2002; Berdin et al., 2003; Krauss et al., 2003; Ellison, 1989, 1993, 1996a, 2005, 2006; Rogers et al., 2005a). Pollen analysis from ocean-floor cores and the presence of mangrove peat across continental shelves document mangrove extent during the Late Quaternary glacial lowstand, providing evidence for mangrove landward retreat with rapid relative sea-level rise during the early and mid Holocene transgression (Woodroffe, 1992; Grindrod et al., 1999; Hanebuth et al., 2000). At mangrove sites which had experienced a rise in site-specific relative sea-level and were observed to have undergone a reduction in area, the landward migration of the wetland margins and reduction in area was likely the result of a combination of site-specific relative sea-level rise and the wetland's physiographic location: the rate of change in elevation of the wetland surface was likely exceeded by the rate of site-specific relative sea-level rise, and the landward mangrove margin migrated landward at a slower rate than did the seaward margin.

In the case where mangroves are experiencing site-specific relative sea-level rise, extensive mangrove landward migration into salt marsh and freshwater wetland habitat has been observed, and may result in a net increase in mangrove area when losses at the seaward margin are exceeded by landward gains (Applegate, 1999; Saintilan and Williams, 1999; Rogers et al., 2005a,b; Lovelock and Ellison, 2007). For instance, in Australia, in areas undergoing regional relative sea-level rise, mangrove transgression has been observed, encroaching into salt marsh (Saintilan and Williams, 1999; Rogers et al., 2005a,b) and freshwater wetlands (Applegate, 1999). In southeastern Australia, where the influence of subsurface processes equal that of sedimentation as a control on the change in elevation of the sediment surface of mangroves (Section 1.5.4), subsidence is occurring due to reduced groundwater inputs associated with El Nino phases, contributing to the site-specific rise in relative sea-level, allowing mangroves to migrate

landward to re-colonize salt marshes (Saintilan and Williams, 1999; Rogers et al., 2005a,b).

#### **1.5.2.2. Wetland Elevation Keeps Pace or Exceeds Rate of Rising Sea-level**

During periods of regional relative sea-level rise, coastal wetlands have also been observed to have maintained or expanded in area (e.g., McCaffery and Thompson, 1980; Pethick, 1980; Woodroffe et al., 1985, 1989; Patrick and DeLaune, 1990; Woodroffe, 1990; Cahoon and Lynch, 1997; Alleng, 1998; Applegate, 1999; Saintilan and Williams, 1999; Rogers Shaw and Ceman, 1999; Shaw and Ceman, 1999; Lucas et al., 2002; Berdin et al., 2003; Krauss et al., 2003; Rogers et al., 2005b; Hashimoto et al., 2006; McKee et al., 2007). Observations of stability in mangrove position indicate no change in site-specific relative sea-level. A trend of lowering in site-specific relative sea-level occurred at mangroves observed to have migrated seaward. Under these circumstances, coastal wetlands may also have expanded laterally if adjacent areas were at a lower elevation than the coastal wetland sediment surface, and these adjacent areas developed hydrologic conditions (duration, depth, and frequency of inundation, and drainage) suitable for coastal wetland establishment (e.g., Snedaker, 1995). In many regions, mangroves began to maintain or expand in area during the mid to late Holocene when the rate of global sea-level rise began to slow (Woodroffe, 1987, 1992, 2002), likely when a threshold was reached resulting in site-specific relative sea-level lowering within the mangrove ecosystems.

#### **1.5.2.3. Factors Controlling Mangrove Responses to Change in Sea-level**

When site-specific relative sea level rise is the predominant factor controlling mangrove position, mangrove responses over decades will generally follow trends shown by paleoenvironmental reconstructions of mangroves to past sea level fluctuations (Section 1.5.2.5) (Woodroffe et al., 1985; Ellison and Stoddart, 1991; Woodroffe, 1995; Shaw and Ceman, 1999; Ellison, 1993, 2000; Berdin et al., 2003). Mangrove vulnerability and responses to relative sea-level rise over human time scales are a result of four main factors (Ellison and Stoddart, 1991; Woodroffe, 1995; Alleng, 1998; Shaw and

Ceman, 1999; Ellison, 1993, 2000; Lucas et al., 2002; Lovelock and Ellison, 2007):

- (i) The contemporary rate of change in site-specific relative sea level, which accounts for all factors affecting the trend in site-specific relative sea-level, including the geomorphic setting, sediment processes (including possible feedbacks to changing hydrology, Section 1.5.5), surface root growth, and subsurface processes;
- (ii) Mangrove species composition;
- (iii) The mangrove's physiographic setting (slope of the land adjacent to the mangrove, slope of the mangrove, and presence of obstacles to landward migration); and
- (iv) Cumulative effects of all stressors.

Rising sea-level will have the greatest impact on mangroves experiencing net lowering in sediment elevation, due to high rates of subsidence exceeding sediment accretion, that are in a physiographic setting that provides limited area for landward migration due to obstacles or steep gradients.

Species composition in individual mangrove systems will influence changes in mangrove structure in response to site-specific relative sea-level rise. Because individual mangrove species have differences in time required to colonize new habitat that becomes available with site-specific relative sea-level rise, the species that colonize more quickly may outcompete slower colonizers and become more dominant (Lovelock and Ellison, 2007). For instance, historically, Rhizophoraceae had greater dominance during periods of past sea-level rise than present (Chappell and Grindrod, 1985; Crowley, 1996; Grindrod et al., 1999; Hashimoto et al., 2006), indicating that Rhizophoraceae may again increase in dominance with projected acceleration in rates of sea-level rise to match rates in the Quaternary and early-mid Holocene. Furthermore, because different mangrove vegetation zones (or mosaics of mangrove communities) have different rates of change in sediment elevation (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007), different vegetation communities may have different degrees of resistance and resilience to rising sea-level.

In addition, the cumulative effects of all stressors on a mangrove site affect its resistance and resilience to sea-level rise. For instance, pollutant

inputs can reduce mangrove productivity, including reducing belowground root production, causing a reduction in the rate of change in elevation of the sediment surface, reducing the system's resistance to regional relative sea-level rise. Or, disturbance by pigs may inhibit mangrove colonization of landward habitat, reducing mangrove ecosystem resistance to regional relative sea-level rise. There may be opportunities to reduce certain physiological stressors (both anthropogenic impacts and natural catastrophic events), which may be reducing local mangrove ecosystem resistance and resilience.

Under various conditions, some mangrove sites will revert to a narrow mangrove fringe, possible survival of individual trees, or even experience local extirpation of the mangrove community (Ellison and Stoddart, 1991; Ellison, 1993, 1996, 2000, 2001, 2006; Woodroffe, 1995; Alleng, 1998; Lucas et al., 2002):

- (i) Where mangrove species do not colonize new habitat at a rate that keeps pace with high rates of relative sea level rise;
- (ii) Where the slope of land upslope from the mangrove is steeper than that of the land the mangrove currently occupies;
- (iii) Where there are obstacles (e.g., seawalls and other erosion control structures) to landward migration of the mangrove landward boundary; and
- (iv) When stressors are reducing mangrove ability to keep pace with regional relative sea-level rise (reduced resistance) and stressors are reducing mangrove ability to colonize or migrate to land at higher elevations (reduced resilience).

All of these aspects have to be favorable for mangrove sites to survive under conditions of rising relative sea level.

The tidal range of a mangrove location influences the response to site-specific relative sea-level rise. For instance, a study in southern Australia documented that sedimentation increased linearly with tidal range in mangroves and salt marshes (Rogers, 2005), suggesting that, if other controls on elevation of the mangrove sediment surface are equal, sites with a larger tidal range are less vulnerable to sea-level rise. Also, a larger proportion of mangrove area will be affected by rising sea-level in microtidal



settings relative to macrotidal settings (Nichols et al., 1999; Lovelock and Ellison, 2007). Mangroves with large tidal ranges tend to have a greater area with the tidal limits and also have a larger number of forest zones (Woodroffe, 1990). For instance, Nicholls et al. (1999) modeled wetland vulnerability to site-specific relative sea-level rise, where vulnerability is directly proportional to the inverse of tidal range, defined as dimensionless relative sea level rise ( $RSLR^* = RSLS / \text{tidal range}$ ) with a critical value ( $RSLR^*_{crit}$ ) above which wetlands will be lost, assuming a complete lack of landward migration. Assuming similar slopes, in areas with small tidal ranges, mangrove systems are generally narrower compared to settings with large tidal ranges, such that a change in site-specific relative sea-level causes a larger proportion of the microtidal wetland to be lost from the seaward edge and gained at the landward margin (Fig. 1.5).

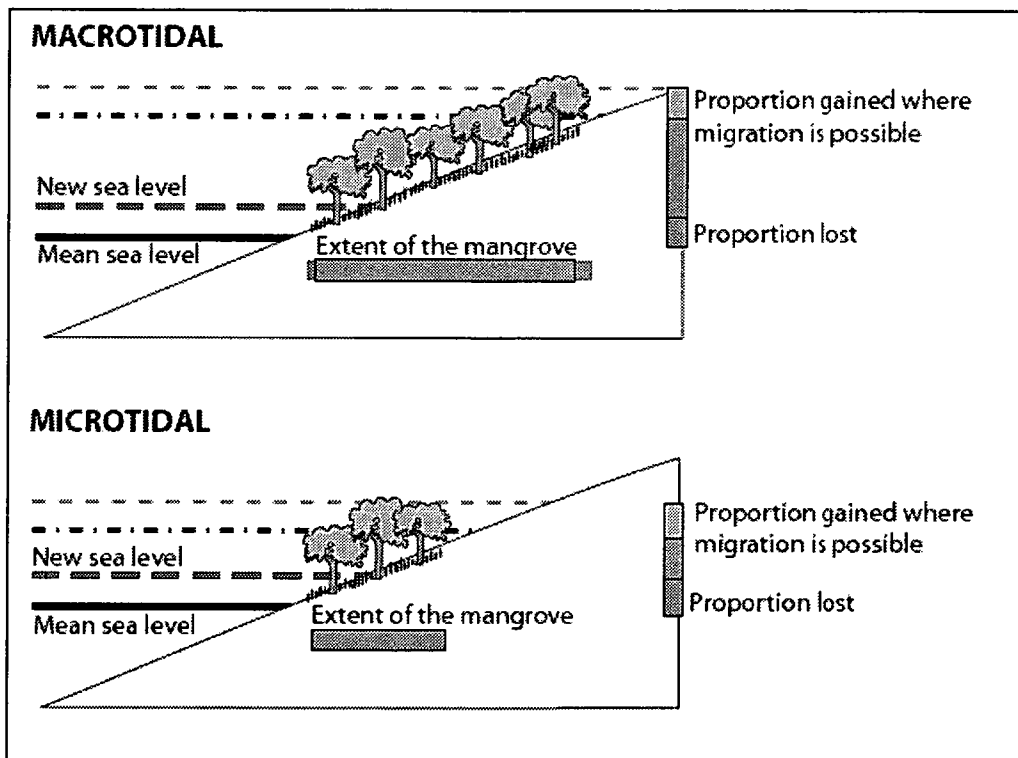


Fig. 1.5. A mangrove's tidal range determines the proportion of a mangrove site that will be affected by change in site-specific relative sea-level (modified from Woodroffe, 1990 by Lovelock and Ellison, 2007).

There are three general scenarios for mangrove responses to trends in relative sea-level, given a time period of decades or longer and where other stressors are relatively small (Fig. 1.6):

- **Stable site-specific relative sea-level:** When sea-level is not changing relative to the mangrove surface, mangrove elevation; salinity; frequency, duration, and depth of inundation; and other factors that determine if a mangrove community can persist at a location will remain relatively constant and the mangrove margins will be relatively stable (Fig. 1.6A) (Blasco, 1996; Alleng, 1998);
- **Site-specific relative sea-level lowering:** When sea-level is falling relative to the mangrove surface, this causes the mangrove margins to migrate seaward (Fig. 1.6B) (Egler, 1952). The mangrove may also expand laterally (Snedaker, 1995) if areas adjacent to the mangrove, which are currently at a lower elevation than the mangrove surface, develop conditions suitable for mangrove establishment; and
- **Site-specific relative sea-level rising:** If sea-level is rising relative to the elevation of the mangrove sediment surface, the mangrove's margins retreat landward as the mangrove species zones migrate inland as they maintain their preferred duration, frequency and depth of inundation (Fig. 1.6C) (Egler, 1952; Semeniuk, 1980; Field, 1995; Ellison, 1993, 2000; Woodroffe, 1995; Lovelock and Ellison, 2007). The mangrove may also expand laterally into areas of higher elevation. Environmental conditions for recruitment and establishment of mangroves in new areas that become available with site-specific relative sea level rise include suitable hydrology and sediment composition of the area inland, competition with non-mangrove plant species and availability of waterborne seedlings.

The seaward mangrove margin migrates landward from mangrove tree dieback due to stresses caused by a rising sea-level such as erosion resulting in weakened root structures and falling of trees, increased salinity, and too high a duration, frequency, and depth of inundation (Naidoo, 1983; Ellison, 1993, 2000, 2006; Lewis, 2005). Mangroves migrate landward via seedling recruitment and vegetative reproduction as new habitat becomes available landward through erosion, inundation, and concomitant change in salinity (Semeniuk, 1994). Depending on the ability of individual mangrove

species (Duke et al., 1998) to colonize newly available habitat at a rate that keeps pace with the rate of relative sea level rise, slope of adjacent land and the presence of obstacles to landward migration of the landward boundary of the mangrove, some mangrove sites will revert to a narrow fringe, possible survival of individual trees, or even experience extirpation (Fig. 1.6D) (Ellison and Stoddart, 1991). Seawalls and other coastal engineering structures, if involved, often lead to serious erosion problems in front of and immediately downstream from the structure (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997).

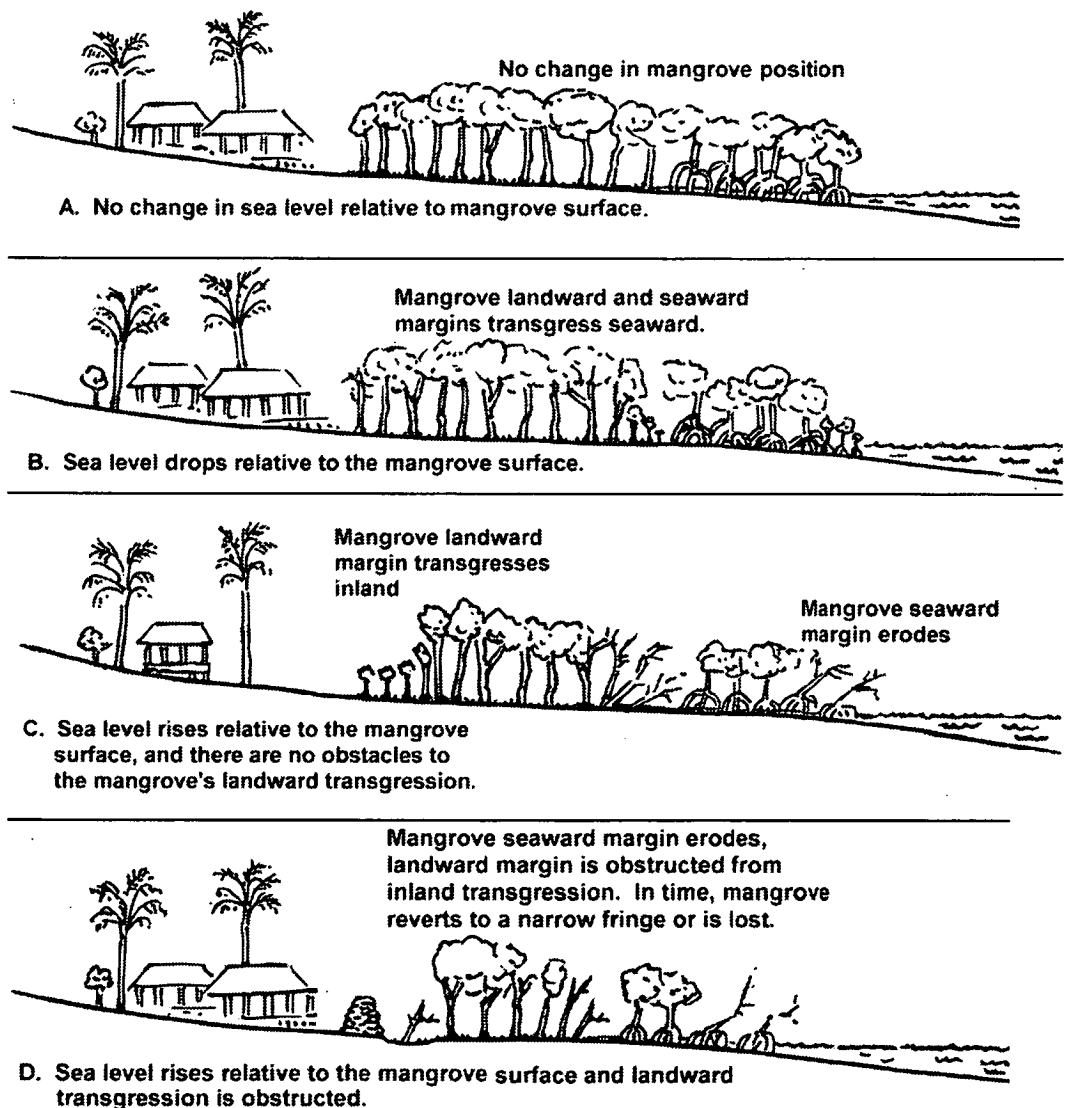


Fig. 1.6. Four scenarios for generalized mangrove response to relative sea-level rise.

#### **1.5.2.4. Sea-level Rise Threat to Coral Reefs**

Increased CO<sub>2</sub> concentrations, concomitant ocean acidification and reduced calcification rates of corals, and temperature changes are believed to be much larger threats to coral reefs compared to relative sea-level rise (Birkeland, 1997; Brown, 1997; LeClerq et al., 2002; Kleypas et al., 1999, 2006;). However, it is worth briefly discussing the state of understanding of coral-reef vulnerability and responses to changes in sea-level as there are some similarities to mangrove and other coastal wetland ecosystems. If sea-level rises at a rate that is slower than the reef's ability to produce carbonate, the reef will prograde seaward as well as aggrade vertically. If the relative sea-level rise rate is roughly equal to the reef's rate of carbonate production, then the reef will grow vertically and not grow seaward or landward. If sea-level outpaces the accreting reef, the reef will either backstep to higher ground or drown (Hubbard, 1997). Reefs may also survive at deeper depths as they grow upward at a lower rate than the rise of sea-level, and catch up if and when the sea-level rise rate slows (Brown, 1997).

Most coral reef communities are expected to be able to keep pace with projected rates of sea-level rise (Birkeland, 1997; Brown, 1997; Wilkinson, 1999; McCarthy et al., 2001). Reef accretion rates range from 1-10 mm a<sup>-1</sup>, with a rate of 10 mm a<sup>-1</sup> accepted as the maximum vertical accretion rate that a reef can sustain (Brown, 1997). However, reef systems may be able to build upward at faster rates, as high as 20 mm a<sup>-1</sup>, when growing in water depths of less than 20 m where there is abundant sunlight for photosynthesis (Brown, 1997). The equivalent median projected global sea-level rise linear rate of the Intergovernmental Panel on Climate Change is 4.3 mm a<sup>-1</sup>, and the maximum projected scenario is the equivalent of a linear rate of 8.0 mm a<sup>-1</sup> from 1990 through 2100 (Church et al., 2001), suggesting that, in general, coral reef communities will be able to keep pace.

However, some reef communities may experience mortality as a result of relative sea-level rise. Reef flat communities that undergo accelerated coral growth to keep pace with rising relative sea-level would become susceptible to subaerial exposure and substantial mortality if sea-level rise occurs in episodic pulses with periods of sea-level remaining steady (Brown, 1997). Also, deeper reefs may not be able to keep pace with projected sea-level rise scenarios. Furthermore, anthropogenic stresses on reef communities,

including increased sedimentation, nutrient loading, rising temperatures, and indirect stresses resulting from the degradation of adjacent coastal communities, are expected to reduce coral reefs' resistance and resilience to accelerated rates of relative sea-level rise (Hubbard, 1997).

#### **1.5.2.5. Paleo-shoreline Reconstructions**

Paleoenvironmental shoreline reconstructions are useful for establishing the long-term response of coastal systems to past sea-level fluctuations (Woodroffe et al., 1985; Sugimura et al., 1988; Ellison and Stoddart, 1991; Woodroffe, 1992, 1995; Shaw and Ceman, 1999; Berdin et al., 2003; Ellison 1993, 2000, 2006). Analysis of the stratigraphy and chronology of Holocene deposits provide insight into how mangrove ecosystems responded to past sea-level rise over thousands of years. This requires that sufficient samples are taken to characterize a site, in particular, to understand dynamics at the seaward and landward margins. For instance, cores taken from strategic locations can document changes in position of the wetland's margins, documenting either stability, wetland transgression, or wetland seaward and possibly lateral progradation. The paleoenvironmental record of mangroves demonstrates mangroves' sensitivity to even small rates of increase in relative sea-level, specifically when there is site-specific relative sea-level rise, ranging from gradual landward movement in cases where the rate of site-specific relative sea-level rise is small, to massive mortality events when the rate is fast. Methods employed to conduct sea-level reconstructions and paleoenvironmental responses of coastal systems include: Analyses of stratigraphy, pollen in sediment, fossil roots and rhizomes, <sup>14</sup>C age dates of plant and animal remains, total organic carbon content, marine terraces, beach ridges, tidal notch measurements, diatom, foraminifera, ostracode, and testate amoebae (e.g. Woodroffe et al., 1985; Scott and Medioli, 1986; Sugimura et al., 1988;; Komar, 1989; Ellison, 1989, 1993, 1998; Shaw and Ceman, 1999; Donnelly and Bertness, 2001; Allen, 2003; Berdin et al., 2003).

During the postglacial marine transgression of the early Holocene (10,000 – 6000 BP), when eustatic sea level rise was rapid, there were no extensive mangrove ecosystems, mangrove ecosystems on mid-oceanic coral atolls were likely nearly extirpated during the period, and on broad, low gradient continental shelves, where the shoreline might have moved inland

on the order of hundreds of kilometers, sea-level rise might have caused a substantial reduction in mangrove area (Ellison and Stoddart, 1991; Woodroffe and Grindrod, 1991; Woodroffe, 1987, 1992). During this period of rapid relative sea-level rise, mangroves persisted in what Ellison and Stoddart (1991) term 'refuge mode', as narrow coastal fringes or scattered individual trees. Mangrove recolonization likely occurred after rates of sea-level rise slowed, when extensive mangroves again became established, at about 6500-4000 BP (Ellison and Stoddart, 1991; Woodroffe and Grindrod, 1991; Woodroffe, 1992). Based on observations of the pollen record in mangrove sediment, distribution of some mangrove species in the Pacific is more restricted today than in the past, likely a result of local extirpations occurring during periods of marine transgressions (Ladd, 1965; Leopold, 1969; Mephram, 1983; Ellison, 2006).

While paleo-shoreline reconstructions provide a basis for understanding the long-term responses to changes in sea-level, these studies may provide unreliable predictions of mangrove responses to changes in sea-level over coming decades. This is due to the relatively recent introduction of anthropogenic activities (Sections 1.3.8, 1.5.3), which in some areas have modified mangrove functioning and structure, including the individual processes that control the elevation of the mangrove sediment surface (e.g., Cahoon and Hensel, 2006) from how they functioned over hundreds and thousands of years in the past. Furthermore, development creates obstacles to natural mangrove migration in response to rising seas, which generally was not a factor in determining paleo-shoreline position.

#### **1.5.2.6. Observations of Trends in Sedimentation and Surface Elevation**

Mangroves of low relief islands in carbonate settings that lack rivers were thought to be the most sensitive to sea-level rise, owing to their sediment-deficit environments (Thom, 1984; Ellison and Stoddart, 1991; Parkinson et al., 1994; Woodroffe, 1987, 1995, 2002). Based on a review of the peat stratigraphic record of mangrove ecosystems during the Holocene, Ellison and Stoddart (1991) predicted that mangroves will be unable to keep pace and maintain current position with a rise in relative sea-level exceeding about  $1.2 \text{ mm a}^{-1}$  at sites with autochthonous organic accumulation and lacking any allochthonous sediment input (on low-islands, in limestone carbonate settings,

where inorganic sedimentation is low), while Parkinson et al. (1994) predicted that Caribbean mangroves in carbonate settings will be unable to maintain position with a rise in site-specific relative sea-level exceeding about 1.3 mm a<sup>-1</sup>. However, more recent studies have shown that subsurface controls on mangrove sediment elevation can offset high or low sedimentation rates (Cahoon et al., 2006; Cahoon and Hensel, 2006), such that sedimentation rates alone provide a poor indicator of vulnerability to rising sea-level.

Information from studies employing the SET (surface elevation table) method (Boumans and Day, 1993) enable a determination of the site-specific change in sea-level relative to the wetland sediment surface (Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Cahoon and Hensel, 2006; McKee et al., 2007). Studies which only quantify sediment accretion and erosion are less reliable in determining if a wetland was keeping pace with regional relative sea-level changes, as they do not account for the full suite of controls on sediment elevation (Section 1.5.4). Several studies using SET technology document large and in some cases significant differences between trends in sediment accretion and trends in sediment elevation in mangroves (e.g., Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon et al. 2006). Furthermore, there have been observations of substantial control of mangrove sediment elevation from groundwater changes primarily from the deepest soil horizon adjacent to bedrock (Whelan et al., 2005). This highlights the need to employ sediment elevation monitoring methods that account for subsurface processes throughout the entire soil profile.

Studies designed to measure trends in mangrove and salt marsh sediment elevation have included about 12 sampling locations per wetland system (range of 6-27) (Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007). Observations of disparate trends in sediment elevation within different vegetation communities of an individual mangrove (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007) support including a substantially larger number of sampling locations within and across different vegetation communities and localized geomorphic settings in order to ensure that a site is adequately characterized.

The mangrove sites where SET-MH technology has been employed, or similar method for which results have been reported, had been monitored for periods of between 1 and 3.6 years (Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007). These study periods may be substantially too short, especially if broad spatial sampling were appropriately incorporated into sampling designs. To conduct an accurate vulnerability assessment of coastal wetlands to changes in site specific relative sea-level, monitoring must be conducted over periods that span the range of variability in surface and subsurface processes that control the elevation of wetland sediment surfaces in order to differentiate between long-term linear trends and cyclical (including seasonal) trends (French and Stoddart, 1992; Kirby et al., 1993; Semeniuk, 1994; List et al., 1997; Reed et al., 1999; Krauss et al., 2003; Cahoon et al., 2006).

#### **1.5.2.7. State of Knowledge and Gaps in Understanding of Mangrove Vulnerability and Responses to Sea-level Rise**

The body of literature reviewed in this section highlights the following lessons learned related to assessing mangrove vulnerability and predicting responses to relative sea-level rise over human time scales:

- **Human-Time-Scale Predictions Require Recent Observations:**  
Paleo-shoreline reconstructions provide a basis for understanding the long-term responses to changes in sea-level. However, they do not provide the resolution needed to understand responses over decades, and the recent influence of anthropogenic activities affects mangrove functioning, structure and migration/colonization of new areas was not a factor until relatively recently. As a result, assessment and monitoring mangrove changes in response to sea-level change and physiographic setting over recent times is necessary to accurately predict responses over coming decades.
- **Monitor entire soil profile:** It is necessary to monitor trends in sediment surface elevation through the entire soil profile. Both surface and subsurface processes can be important controls on mangrove sediment surface elevation. Sedimentation rates alone provide a poor indicator of vulnerability to rising sea-level.



- **Adequate spatial sampling:** Observations of disparate trends in sediment elevation within a mangrove site highlight the importance of employing spatial sampling to adequately characterize entire sites.

There are several important gaps in the state of understanding mangrove vulnerability and responses to sea-level rise:

- **Comprehensive assessment of vulnerability and responses:** There have been no reported comprehensive site-based assessments of both mangrove vulnerability and predicted responses to relative sea-level rise. Studies observing changes in mangrove position or trends in sedimentation or surface elevation have not also assessed the sites' capacity to naturally migrate landward, or other predicted changes in response to rising seas.
- **Study designs to monitor trends in mangrove elevation:** Spatial sampling designs for monitoring trends in sediment surface elevation have likely been inadequate to adequately characterize entire sites. Furthermore, studies monitoring trends in mangrove sediment surface elevation have only been recently initiated, and have relatively short study periods that may inadequately differentiate between cyclical and linear trends. As a result, the threat to mangroves from relative sea-level rise is not well understood. Improved characterization of sites and longer study periods are needed.
- **Mangrove vulnerability – global or regional phenomena:** Observations of trends in surface elevations are primarily from mangroves of the western Pacific and wider Caribbean regions. Assessments in other regions are needed to determine if the preliminary determination that most mangrove sites are not keeping pace with relative sea-level rise is a global vs. regional phenomenon.
- **Predictive elevation models:** There is a lack of reliable predictive models for trends in mangrove sediment surface elevation, and lack of documentation of correlations between geomorphic class and sediment elevation trends (Section 1.5.1). As a result, site-specific assessment is required.
- **Predictive erosion models:** There are no reported observations relating mangrove margin movement to relative sea-level change. There is a lack

of reliable predictive models for trends in mangrove margin position in response to changes in sea-level.

### **1.5.3. Non Climate Change Stressors**

Non climate change-related factors that stress mangrove ecosystems, including those caused by human activities, can affect a mangrove's resistance and resilience to the additional stress of sea-level rise and other climate change outcomes. Coastal wetland changes in position over thousands of years before present, when human influences were minimal, were likely caused primarily by the rate of change in site-specific relative sea-level. However, over human time scales, the vulnerability of mangroves to site-specific changes in relative sea-level, and the threshold rate of rise in site-specific relative sea-level that an individual mangrove can withstand before changes in position occurs, is now also heavily influenced by anthropogenic-driven stressors, including human responses to climate change.

Mangrove species have specific tolerance levels for the hydrologic and salinity regimes; wave energy; soil and water pH; sediment composition and stability; nutrient concentrations; and degree of faunal predation. This determines if a mangrove can establish and survive in a specific location and results in zonal distribution of mangrove species (Tomlinson, 1986; Naidoo, 1985, 1990; Wakushima et al., 1994a, 1994b; Duke et al., 1998; Duke, 1992). While there is still incomplete understanding of what combination of factors control mangrove establishment and health, changes in any of these factors can result in changes in the location of mangrove margins (Duke et al., 1998; Donnelly and Bertness, 2001; Saintilan and Wilton, 2001; Wilton, 2002).

Factors affecting mangrove sediment-budget balances, most of which are influenced by human activities, include changes in sediment inputs, variations in coastal currents and wind directions and strength, variations in regional climate and resulting storms, and construction of seawalls, other shoreline erosion control structures, and other structures that prevent the landward migration of coastal wetlands with site-specific relative sea-level rise. These influences on mangrove sedimentation processes affect mangrove responses to site-specific relative sea-level rise.

In addition to altered sediment inputs, several other non-climate-change-related factors can affect mangrove margin position, as well as structure and health. These include disrupting connectivity to adjacent ecosystems; changing nutrient, surface freshwater, and pollutant inputs; extracting groundwater; clearing mangrove vegetation; filling; displacing native species with alien invasive species; and harming vegetation from insect infestations, fungal flora pathogens, and other diseases (United Nations Environment Programme, 1994; Ellison, 1993, 1996, 1999; Gilman, 1999a, 1999b; Donnelly and Bertness, 2001; Saintilan and Wilton, 2001; Krauss et al., 2003; Lovelock and Ellison, 2007). Factors that may cause mass mangrove mortality (both from natural processes and anthropogenic activities) include excessive sedimentation (Lugo and Cintron, 1975; Hutchings and Saenger, 1987; Ellison, In Press), hydrological blockage causing either sustained or restricted inundation (Hatton and Couto, 1992), sediment erosion, oil spills (Lewis, 1983; Duke et al. 1997), and clearing (Diop, 2003).

To predict site-specific mangrove responses to regional relative sea-level rise, it is necessary to determine if the change in sea-level is the predominant control over mangrove position, as well as structure and health, or if other stressors are predominant controls. Observation of a significant positive correlation between a change in relative sea level and change in position of mangrove margins has been used to support the inference that change in site-specific relative sea level is the predominant influence in determining the mangrove margin positions (Saintilan and Wilton, 2001; Wilton, 2002).

#### **1.5.4. Controls on Mangrove Sediment Surface Elevation**

When the rate of change in elevation of a mangrove's sediment surface is exceeded by the rate of change in relative sea-level, the mangrove is not keeping pace with rising seas. There are several interconnected processes that influence the elevation of mangroves' sediment surface (Fig. 1.7, Table 1.3).

Hydrology directly affects wetland elevation through processes of compression and dilation storage (Cahoon et al., 2006). Water storage in wetland soils is controlled by: (i) 'saturation storage', where, in the sediment

horizons above the water table, pore space can either be occupied by gases or water; and (ii) 'dilation storage' or 'shrink-swell', where throughout the wetland sediment, the more water that is incorporated into the sediment below the water table, the more the sediment dilates, increasing sediment volume, increasing the elevation of the wetland sediment surface (Cahoon et al., 2006). The amount of dilation storage and degree of change in elevation of the sediment surface varies with soil type. Changes in groundwater inputs, such as from long-term changes in precipitation levels resulting from global climate change, would result in a long term (vs. seasonal) change in mangrove elevation. Short-term cyclical influences include variability in precipitation and tidal range. Research has demonstrated the short-term effects of groundwater recharge on mangrove elevation: Rogers et al. (2005) observed a direct correlation between change in mangrove surface elevation and monthly rainfall levels during a severe El Nino drought phase in a mangrove in eastern Australia, while Whelan et al. (2005) found a direct correlation between mangrove surface elevation and seasonal variability in groundwater pressure in a Florida mangrove. Research is lacking to demonstrate effects of long term trends in changes in groundwater inputs.

Table 1.3. Processes known to control the elevation of mangrove sediment surfaces.

Process	Influence on Mangrove Sediment Surface Elevation
Sediment accretion and erosion	Sediment accretion and erosion are determined by a mangrove's geomorphic setting, which affects the sources of sediment, sediment composition, and method of delivery (Furukawa and Wolanski, 1996; Furukawa et al., 1997; Woodroffe, 1990, 2002). Fine sediment particles are carried in suspension into mangrove systems from coastal waters during tidal inundation, form large flocs (cohesive clay and fine silt), which settle in the forest during slack high tide as the friction caused by the high mangrove vegetation density slows tidal currents. Wrack or plant litter on the soil surface can also trap mineral sediment, and contribute to vertical accretion (Cahoon et al., 2006). Water currents during ebb tides are too low to re-entrain the sediment. Thus, the mangrove structure causes sediment accumulation (Furukawa and Wolanski, 1996). Storms can alter the mangrove sediment elevation through soil erosion and deposition (Cahoon et al., 2003, 2006). Sedimentation varies by mangrove species and their root type (Furukawa and Wolanski, 1996; Krauss et al., 2003).
Biotic contributions	Biotic contributions to soil elevation vary from low (allochthonous mineral soils) to very high (autochthonous peat soils), where surface processes include accumulation of decaying organic matter such as leaf litter, and formation of living benthic microbial, algal or root mats (Woodroffe, 1992, 2002; Cahoon et al., 2006). The accumulation of leaf litter is controlled by aboveground production, detritivore consumption, microbial decomposition and tidal flushing (Middleton and McKee, 2001; Cahoon et al., 2006).
Belowground primary production	When belowground root growth exceeds root decomposition, soil organic matter accumulates, causing a net increase in soil volume and contributes to a rise in sediment elevation. Root growth, or the lack thereof, has been shown to be a substantial control on mangrove soil elevation at some sites (Cahoon et al., 2003, 2006; Cahoon and Hensel, 2006; McKee et al., 2007). In particular, mangroves in carbonate settings, such as on low oceanic islands remote from continental sources of sediment, have autochthonous soil, composed primarily of mangrove roots, where belowground primary productivity and organic matter accumulation are the primary controls on sediment elevation (Cahoon et al., 2006; McKee et al., 2007).
Auto-compaction	Autocompaction, the lowering of the sediment surface and reduction in sediment volume, is caused by the oxidation (decomposition) and compression of organic material, and inorganic processes, including rearrangement of the mineral architecture, silica solution, clay dehydration and other diagenetic processes (Kaye and Barghoorn, 1964; Pizzuto and Schwendt, 1997; Cahoon et al., 1995, 1999; Allen, 2000; Woodroffe, 2002; Cahoon and Hensel, 2006). Autocompaction is understood to decrease asymptotically with the age of the mangrove (Woodroffe, 2002). Mangroves suffering mass tree mortality, caused by storms or other acute sources of stress, at sites where the substrate is composed primarily of peat or organic mud, are susceptible to substantial lowering in elevation of their sediment surface through peat collapse and soil compression (e.g., Cahoon et al., 2003).
Fluctuations in water table levels and pore water storage	Hydrology directly affects wetland elevation through processes of compression and dilation storage (Cahoon et al., 2006). The more water that is incorporated into the sediment below the water table, referred to as 'dilation storage' or 'shrink-swell', the more the sediment dilates, increasing sediment volume, increasing the elevation of the wetland sediment surface (Cahoon et al., 2006). The amount of dilation storage and degree of change in elevation of the sediment surface varies with soil type. Changes in groundwater inputs, such as from long-term changes in precipitation levels resulting from climate change, would result in a long term change in mangrove elevation. Short-term cyclical influences include variability in precipitation and tidal range. Research conducted to date has demonstrated the short-term effects of groundwater recharge on mangrove elevation (Rogers et al., 2005; Whelan et al., 2005). Research is lacking to demonstrate effects of long term trends in changes in groundwater inputs.

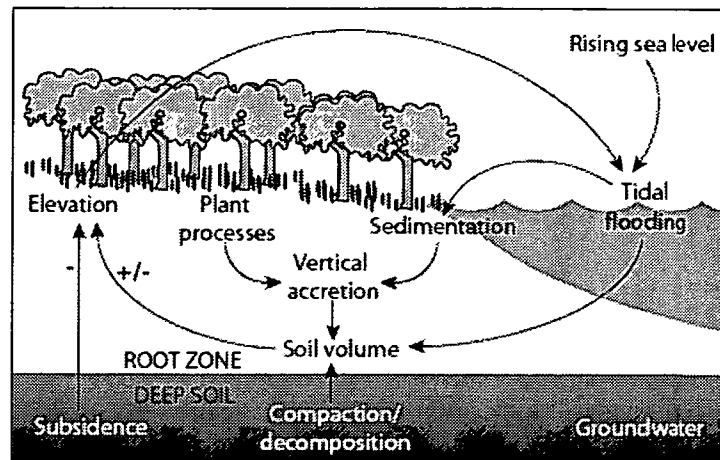


Fig. 1.7. Model of the interconnected processes and factors believed to be primary controls on changes in the elevation of the mangrove sediment surface (adapted from Cahoon et al., 1999 by Diane Kleine).

Autocompaction, the lowering of the sediment surface and reduction in sediment volume, is caused by the oxidation (decomposition) and compression of organic material, and inorganic processes, including rearrangement of the mineral architecture, silica solution, clay dehydration and other diagenetic processes (Kaye and Barghoorn, 1964; Pizzuto and Schwendt, 1997; Cahoon et al., 1995, 1999; Allen, 2000; Woodroffe, 2002; Cahoon and Hensel, 2006). Autocompaction is understood to decrease asymptotically with the age of the mangrove (Woodroffe, 2002).

Sediment accretion and erosion are determined by a mangrove's geomorphic setting, which affects the sources of sediment, sediment composition and method of delivery (Furukawa and Wolanski, 1996; Furukawa et al., 1997; Woodroffe, 1990, 2002). Mangrove sedimentation is generally controlled by a mangrove's hydroperiod (duration, frequency and depth of inundation), currents, tidal and wave energy, degree of flushing (hydrologic turnover), storms (pulses of sediment deposition and erosion), influence of rivers, tidal creek system, and organic matter production, deposition, decomposition, and export (Furukawa and Wolanski, 1996; Furukawa et al., 1997; Cahoon et al., 1999; Woodroffe, 1995, 2002). For instance, river-dominated systems receive an allochthonous sediment supply, where the volume of sediment deposition is generally a function of the catchment size, organic production is augmented by the freshwater input,

and this organic matter is exported through river flow (Fleming et al., 1990; Furukawa et al., 1997; Woodroffe, 1990, 2002). Tide-dominated systems (fringe and overwash classes, Cintron et al. [1985] categorization) also contain abundant allochthonous sediment but the tides are the primary factor controlling sediment redistribution, there is bi-directional flux, transporting organic matter from the mangrove to adjacent coastal areas, but to a more limited extent than in riverine settings (Jiminez and Sauter, 1991; Lee, 1995, 1999; Woodroffe, 1990, 2002). Finally, interior/basin mangroves in depressional settings are generally remote from direct flushing, are sinks for organic matter, nutrients and sediment, and export relatively little sediment or organic matter (Twilley, 1985; Woodroffe, 2002).

Fine sediment particles are carried in suspension into mangrove systems from coastal waters during tidal inundation, form large flocs (cohesive clay and fine silt), which settle in the forest during slack high tide as the friction caused by the high mangrove vegetation density slows tidal currents. Wrack or plant litter on the soil surface can also trap mineral sediment and contribute to vertical accretion (Cahoon et al., 2006). Water currents during ebb tides are too small to re-entrain the sediment. Thus, the mangrove structure causes sediment accumulation (Furukawa and Wolanski, 1996). Sedimentation is largest at mangrove trees with a complex root structure, such as *Rhizophora* sp., and is smallest for single trees such as *Ceriops* sp. (Furukawa and Wolanski, 1996). Krauss et al. (2003) found that the rate of change of mangrove sediment surface elevation varied from  $-0.2 \text{ mm a}^{-1}$  to  $3.4 \text{ mm a}^{-1}$  according to the mangrove species and their root type.

Biotic contributions to soil elevation vary from low (allochthonous mineral soils) to very high (autochthonous peat soils), where surface processes include the accumulation of decaying organic matter, such as leaf litter, and the formation of living benthic microbial, algal or root mats (Woodroffe, 1992, 2002; Cahoon et al., 2006). The accumulation of leaf litter is controlled by aboveground production, consumption by detritivores, microbial decomposition and tidal flushing (Middleton and McKee, 2001; Cahoon et al., 2006). For instance, vertical accretion in interior basin mangroves is likely to be controlled primarily by organic matter accumulation, while in fringe mangroves, with more frequent tidal flushing and faster decay of organic matter, accumulated organic matter is not likely to contribute

substantially to vertical accretion (Woodroffe, 2002; Cahoon et al., 2006). Systems where both terrestrial and calcareous sediment supply is low or absent are autochthonous, i.e. sediments are produced primarily within the wetland ecosystem. For instance, mangroves of oceanic islands, coral reefs and carbonate banks typically are in carbonate settings with a small tidal range and where the autochthonous mangrove sediment is made of peat (Woodroffe, 1992).

Mangroves suffering mass tree mortality, caused by storms or other acute sources of stress, at sites where the substrate is composed primarily of peat or organic mud, are susceptible to substantial lowering in elevation of their sediment surface (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Woodroffe, 1995b, 2002; Cahoon et al., 2003). Storms can alter the mangrove sediment elevation through soil erosion, soil deposition, peat collapse, and soil compression (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Smith et al., 1994; Mastaller, 1996; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Baldwin et al., 2001; Sherman et al., 2001; Woodroffe, 1995b, 2002; Cahoon et al., 2003, 2006; Cahoon, 2006; Piou et al., 2006). For example, an observed loss in mangrove surface elevation at a site in Honduras was believed to be caused by the cessation of root growth combined with peat collapse driven by decomposition, following mass tree mortality from hurricane damage (Cahoon et al., 2003). Results from modeling mangrove peat collapse following the mass mortality of trees from storm damage indicated that areas that do not experience renewed root growth will experience peat collapse for at least a 10 year period following the storm event (Cahoon et al., 2003). A storm surge was inferred to cause 33 mm of sediment compression in a deteriorating salt marsh, which had not rebounded after 8 years (Rybczyk and Cahoon, 2002), which suggests that storm surges could similarly cause surface elevation lowering in deteriorating mangroves. Basin mangroves that suffered mass tree mortality experienced lowering of their sediment surface at a rate of  $11 \text{ mm a}^{-1}$  due to decomposition of dead roots and sediment compaction, while mangroves that experienced minimal storm damage experienced a gain in elevation due just to root production of  $5 \text{ mm a}^{-1}$ , which exceeded the sediment accretion rate



( $2 \text{ mm a}^{-1}$ ) (Cahoon et al, 2003). Sherman et al. (2000) and Whelan (2005) observed similar elevation loss at sites where lightning strikes caused short-term canopy gaps and concomitant short-term (7-10 year) death of tree roots. These observations (Sherman et al., 2000; Cahoon et al., 2003; Whelan, 2005) are consistent with the findings of Cahoon and Hensel (2006), that subsurface processes are primary controls on mangrove elevation, and also highlight the natural threat to mangroves from storm events.

Numerous anthropogenic activities can also alter the mangrove sediment elevation. Groundwater extraction causes lowering of the mangrove sediment surface. Seawalls and other erosion control structures along the mangrove landward margin cause erosion and scouring of the mangrove immediately fronting and down-current from the structure (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997). Sedimentation processes can be altered through increased erosion in a mangrove's catchment and deposition of dredge spoils. Nutrient enrichment has different effects on elevation change depending on the tree species and nutrient added (Feller et al., 2003; McKee et al., 2002, 2007). Nutrient inputs affect mangrove productivity, changing root production and organic material inputs, changing the rate of change in sediment elevation.

Studies that relied only on sedimentation rates or subsurface processes through only a portion of the mangrove sediment profile to determine if a mangrove kept pace with regional relative sea-level changes likely produced unreliable results. As discussed previously, mangroves of low relief islands in carbonate settings that lack rivers were thought to be the most sensitive to sea-level rise, owing to their sediment-deficit environments (Thom, 1984; Ellison and Stoddart, 1991; Woodroffe, 1987, 1995, 2002). However, recent studies document that subsurface controls on mangrove sediment elevation can offset high or low sedimentation rates (Cahoon et al., 2006; Cahoon and Hensel, 2006). As a result, information on sedimentation rates alone is an inadequate indicator of mangrove vulnerability to changes in relative sea-level.

The SET-MH method permits quantifying the contributions of the various surface and subsurface factors that control mangrove sediment elevation. Several studies using SET-MH technology document large and in some cases significant differences between trends in sediment accretion and

trends in sediment elevation in mangroves (e.g., Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon et al. 2006). Furthermore, Cahoon and Hensel (2006) found that subsurface processes, and not sedimentation processes, were primary controlling factors of elevation change at 28 mangrove sites with SET-MH technology. Observation of substantial control of mangrove sediment elevation from groundwater changes (shrink and swell) primarily from the deepest soil horizon adjacent to bedrock (Whelan et al., 2005) highlights the need to employ sediment elevation monitoring methods that account for subsurface processes throughout the entire soil profile. These findings also indicate that observations of vertical sediment accretion and erosion rates are likely to be inadequate to assess vulnerability to site-specific relative sea-level rise, as conducted in numerous previous studies, which compared rates of vertical accretion to regional relative sea-level rise rates to determine if the rate of vertical accretion was exceeded by the rate of regional relative sea-level rise (referred to as an 'accretion deficit') (e.g., Baumann et al., 1984; van de Plassche, 1986; Stevenson et al., 1986; Reed, 1990; Allen, 1990a, 1991; Reed and Cahoon, 1993).

All of the controls on mangrove elevation (Table 1.3) have the potential to be variable within different areas of a mangrove. There have been observations of disparate trends in sediment elevation within an individual mangrove (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007). This highlights the importance of designing sampling methods to observe trends in change in surface elevation to adequately characterize an entire mangrove site, within and between different vegetation communities.

There has been no positive correlation observed between mangrove sediment elevation change and regional relative sea-level rise, tidal range, or soil bulk density, nor are there any broad general positive correlations between geomorphic classes and trends in mangrove sediment elevation (Cahoon and Hensel, 2006). Site-specific monitoring is necessary to determine mangrove vulnerability to relative sea-level rise.

Several of the studies discussed in Section 1.5.2 and included in Table 1.2 demonstrate techniques to separate the contributions of some of the factors controlling coastal wetland sediment elevation, including studies employing the SET-MH method (Cahoon and Lynch, 1997; Krauss et al,

2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007). Information on trends in mangrove sediment elevation alone provides needed information to determine resistance to projected regional relative sea-level rise. However, the additional information of the relative contributions of individual surface and subsurface factors controlling elevation can provide a better educated premise for predicting site-specific mangrove vulnerability. For example, if groundwater recharge has been identified as a primary control on a mangrove's sediment surface elevation over recent years, and increased groundwater extraction in the catchment is predicted over coming decades along with projections for reduced regional precipitation, this would support a prediction for lowering in mangrove sediment elevation. Furthermore, identifying the relevant contributions of individual factors controlling sediment elevation may enable the identification of effective adaptation approaches to reduce and offset predicted mangrove losses. For instance, if a positive correlation was observed between the volume of discharge of nutrients in agricultural runoff into a mangrove and trend in reduced sediment elevation from a reduction in mangrove tree productivity, and this provided strong evidence to infer that the reduced productivity was caused by the nutrient inputs, this would highlight the potential for reducing nutrient inputs into the mangrove as a means to reduce the mangrove's vulnerability to regional relative sea-level rise.

Along coastlines where post-glacial subsidence occurred during the Late Holocene, vertical saltmarsh accretion has been used as a surrogate for the rate of sea-level rise (van de Plassche, 1986; Reed, 1990; Allen, 1990a, 1991). The underlying assumption is that, within certain environmental limits, which are not well defined, the coastal marsh elevation maintains equilibrium with sea-level through feedbacks between biotic and abiotic processes, where the wetland sediment surface is at an elevation determined by sea-level and tidal regime (Allen and Rae, 1988; Nyman et al., 1993, 1995). 'Mature' salt marshes will exhibit an equilibrium level related to tidal parameters, and thus exhibit this response to a rise in site-specific relative sea-level, while 'immature' marshes located at lower positions in the tidal range will show variable rates of accretion and change in elevation of the sediment surface as their surface elevation increases towards a 'mature'

level (Allen, 1991; Cahoon et al., 2006). However, accretion rates have been shown to be a poor indicator of the site-specific sea-level rise rate in coastal wetlands because the elevation of the sediment surface of a coastal wetland is determined by several factors, only one of which is rates of sediment accretion or erosion, where other controlling factors on wetland surface elevation include vertical land movement, anthropogenic influences, and subsurface processes (Lynch et al., 1989; Allen, 1990a; French, 1991; Donnelly and Bertness, 2001; Haslett et al., 2001; Krauss et al., 2003; Rogers et al., 2005a,b; Cahoon et al., 1999, 2006; Cahoon and Hensel, 2006).

#### **1.5.5. Feedback Mechanisms**

This section discusses the state of understanding of feedback mechanisms to changes in relative sea-level that affect processes controlling mangrove sediment surface elevation. The understanding of how surface and subsurface processes that control a mangrove's sediment surface elevation respond to changes in relative sea-level is poor. There are likely several feedback loops, where processes that affect mangrove surface elevation interact with local changes in sea-level. Predictive elevation models used to estimate salt marsh elevation responses to projected sea-level rise (Allen, 1990a, 1992; French, 1991, 1993; Morris et al., 2002; Rybczyk and Cahoon, 2002) have not been developed for mangroves, although Pernetta and Osborne (1988) modeled the landward retreat of mangroves in the Gulf of Papua with a rise in regional relative sea-level. The salt-marsh models developed by Allen (1990a, 1992) and French (1991, 1993) employed an exponentially decreasing rate of inorganic sediment accretion as the elevation of the sediment surface increases, presumably due to decreased tidal inundation frequency and duration. Woodroffe (2002) hypothesized that mangrove sedimentation processes are likely to be similar to those of saltmarsh responses to decreased tidal inundation. The saltmarsh models assume that the rate of organic accumulation resulting from plant production within the marsh is near-constant and not effected by a change in elevation of the sediment surface, and generally ignore possible effects of change in sediment surface elevation on subsurface processes, which has

subsequently been shown to be a poor assumption (Cahoon et al., 1999; Cahoon and Hensel, 2006).

Cahoon and Hensel (2006) and Cahoon et al. (2006), through a review of results from 28 mangrove sites with SET-MH technology, found that there was no significant positive relationship between change in sediment elevation and change in relative sea level, or between elevation change and tidal range, but did find that vertical sediment accretion increased in areas experiencing relative sea level rise, where the relationship was highly significant ( $P < 0.0001$ ,  $N = 41$ ). They found a significant positive linear correlation between accretion and relative sea level rise, as well as a positive relationship between accretion and tidal range, but only for mangroves in estuarine and embayment geomorphic settings (Cahoon and Hensel, 2006). Most importantly for understanding mangrove responses to regional relative sea-level rise, Cahoon and Hensel (2006) found that the majority of the study sites are experiencing a rise in site-specific relative sea-level, and concluded that subsurface processes are primary controls on mangrove elevation in many sites, being more important controls than sedimentation processes.

It is possible that, in carbonate settings, there is a feedback process between mangrove root accumulation and site-specific sea-level. Site-specific sea-level rise may alter mangrove production and decomposition processes, resulting in increased root production and increased rate of change in the elevation of the sediment surface (McKee et al., 2007).

Results reported by Saad et al. (1999) indicate that sediment accretion rates decrease when moving from the seaward margin inland, as the sediment surface increases in elevation. It is possible that the increase in flooding frequency and duration of inundation with relative sea-level rise causes increased sediment deposition, explaining evidence for increased accretion rates at sites experiencing a rise in relative sea-level (Allen, 1990a; Cahoon and Hensel, 2006). If this hypothesis is correct, then we would expect to also observe decreased accretion rates at sites experiencing site-specific relative sea-level lowering (Allen, 1990a).

If sediment accretion increased with increased hydroperiod (duration, frequency and depth of inundation), as increased sedimentation can increase mangrove plant growth by direct effects on elevation as well as increased nutrient delivery, this might further increase sediment accretion through

organic matter deposition as well as enhanced sediment retention with the reduced rate of flow of floodwaters that would occur with higher tree productivity (Cahoon et al., 1999). This would be a negative feedback loop, as the increased sedimentation, and concomitant rise in elevation of the mangrove sediment surface, resulting from increased hydroperiod, would cause a decrease in hydroperiod. Furthermore, increased hydroperiod may increase the mangrove substrate pore water storage (Cahoon et al., 1999), contributing to a rise in elevation of the sediment surface, acting to reduce the hydroperiod.

Allen (1990a) hypothesized that the older the age and 'maturity' of a coastal marsh and shorter the time period of observation of accretion rates, the smaller the deviation between rates of accretion and relative sea-level will be, as these conditions (i) avoid sedimentation processes of 'immature' still-growing marshes, which result in over-estimates of the rate of relative sea-level rise, and (ii) maximize the likelihood of there being a linear trend in relative sea-level. This is based on the hypothesis that saltmarsh vertical growth is a function of the rate of sedimentation of mineral sediment, rate of organic accumulation, rate of change of relative sea-level, and rate of long-term sediment compaction from loading (versus seasonal and short-term effects of drying). The model developed by Allen (1990a) further assumes that the mineral sedimentation rate decreases as the marsh elevation rises, as a result of there being less frequent tidal inundation and shorter duration of inundation. Organic accumulation is presumed to only weakly increase with elevation, because organic sediment can continue to be contributed even after the marsh emerges above tidal influence, as long as groundwater conditions are adequate. Similarly, sediment compaction is hypothesized to increase, but again only slightly, with increased elevation of the marsh sediment surface (Allen, 1990a). Several studies support the hypothesis of reduced sedimentation with marsh maturation, where the older marshes are understood to have reached a higher elevation, reducing the frequency of tidal inundation, and as a result are no longer accreting as rapidly (Pethick, 1980, 1981; Stoddart et al., 1989; French, 1993). Cahoon et al. (2006) found that there was a highly significant positive correlation between accretion rates and rates of regional relative sea-level rise in salt marshes from 108 study sites in North America, Europe and Australia and

also in mangroves from 41 study sites in the Gulf of Mexico, Caribbean, Central America, the western Pacific and Australia, where SET technology was employed. These results indicate that, in general, salt marsh and mangrove surface processes exhibit a sediment accretion feedback loop that contributes to keeping pace with sea-level rise.

French (1991) developed numerical models to simulate how nine coastal marshes will respond to projected eustatic sea-level rise, taking into consideration rates of subsidence, sedimentation rates, rate of projected eustatic sea-level rise and concomitant effects on tidal levels and inundation frequency. Model results suggested that the marshes would persist and keep pace with projections for rise in eustatic sea-level. French (1991) assumed that, under conditions of stable change in eustatic sea-level, that marsh sediment accretion is offset by subsidence at marsh sites that are stable in position, thus assuming a state of equilibrium between sediment accretion and subsidence. At stable coastal marsh sites experiencing eustatic rise in sea-level, marsh sedimentation is assumed to balance both subsidence and rise in sea-level. Increased sedimentation rates are presumed to result with increased inundation frequency. A potential problem with the model is that several subsurface controls on marsh sediment surface elevation were not included, in some cases, because they were assumed to result in nominal influence on elevation, however, more recent research has demonstrated the important influence of these other subsurface processes on coastal wetland elevation (e.g. Cahoon et al., 1999; Krauss et al., 2003; Cahoon and Hensel, 2006).

In conclusion, the state of understanding of feedback mechanisms from change in site-specific relative sea-level that affect processes controlling mangrove sediment elevation is poor. Relatively short-term observations, over periods of a few years, have consistently documented positive correlations between regional relative sea-level rise and sediment accretion, a positive feedback mechanism, which contributes to mangroves keeping pace with regional relative sea-level rise. It is unclear how strong the feedback mechanism is, which is likely site-specific depending on the geomorphic setting and resulting sedimentation processes. Observations over longer time periods of decades and longer and from numerous sites in

settings experiencing rise, lowering and stability in regional relative sea-level, may improve the understanding of this and other feedback mechanisms.

#### **1.5.6. Measuring Trends in Site Specific Relative Sea-Level in Mangroves**

Previous sections have described the need for information on trends in the elevation of mangrove sediment surfaces and trends in sea-level relative to the elevation of the mangrove sediment surface in order to assess the threat to mangrove sites from changes in relative sea-level. A precise method to measure the change in sea-level relative to the elevation of the mangrove sediment surface would be to install a tide gauge within the mangrove site and survey from the tide gauge to a series of surveyed elevations of the mangrove surface relative to benchmarks throughout the site. This would be expensive, in part because it would be labor-intensive to survey across a mangrove site, and would require a minimum of a 20-year tide gauge record to obtain an accurate trend in relative sea level (Church et al., 2004a). This method is also highly vulnerable to human disturbance of the tide gauge or benchmarks.

Alternatively, to accurately assess mangrove site vulnerability to relative sea level rise, information on trends in the change in elevation of the mangrove surface over recent decades can be collected to determine how sea level has been changing relative to the mangrove surface. GPS technology may achieve centimeter-millimeter precision and accuracy for vertical measurements over distances of a few kilometers, but, unfortunately, dense mangrove canopy cover results in changing satellite coverage (loss of lock and reduced satellite availability), making current GPS technology unsuitable to monitor trends in the elevation of mangrove surfaces at the millimeter per year level.

The surface elevation table – marker horizon (SET-MH) method provides high resolution, site specific information on mangrove accretion, change in elevation of the mangrove sediment surface and shallow subsidence (Fig. 1.8) (Boumans and Day, 1993; Cahoon et al., 2002a,b; Cahoon and Hensel, 2006). The SET-MH method simultaneously measures vertical accretion from artificial soil marker horizons, and sediment surface elevation change from a surface elevation table, producing high resolution, 1-



2 mm, observations of vertical change (Boumans and Day, 1993; Cahoon et al., 2002a,b; Cahoon and Hensel, 2006). The surface elevation table rods can be tied into a benchmark, providing a reference point for the mangrove surface elevations. A benefit of employing the marker horizon method in combination with the surface elevation table method is that this enables separation of the contribution of surface sediment accretion and erosion and surface root growth from subsurface processes in causing the observed change in surface elevation. Furthermore, shallow subsidence or expansion can be calculated over different depths of the soil profile (Fig. 1.8) (Cahoon and Hensel, 2006). The shallow rod SET, when used alone, provides information on the site-specific influence of live root zone processes on elevation (Cahoon and Hensel, 2006). This information is useful to understand fundamental mangrove processes, but is not needed to assess mangrove vulnerability to changes in relative sea-level. However, as discussed previously, understanding the amount and direction of surface and subsurface controls on elevation of the mangrove sediment surface may assist in identifying effective management approaches to mitigate predicted mangrove responses to relative sea level rise (Cahoon and Hensel, 2006).

One limitation of the SET-MH method is the need for study sites to be located proximate to a tide gauge with a long (> 20 year) record. However, for sites with a local tide gauge record of < 20 years or that are located far from the nearest tide gauge, sea level trends can be accurately calculated using satellite altimetry data combined with historical global tide gauge records (Church et al., 2004). Furthermore, a large number of sampling locations would need to be included to adequately characterize a mangrove area, as changes in sediment surface elevation have been shown to be variable across a single mangrove site (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007). This can be labor-intensive and expensive, all of the rods should be inserted to depths where they hit consolidated basal sediment if the method is to account for all subsurface processes that may represent substantial controls on the elevation of the sediment surface, which can be time consuming in sites with deep mangrove peat and organic mud layers, and there is a risk of human disturbance of study sites, which is problematic for sites where only a few stations are installed due to budget or staff constraints.

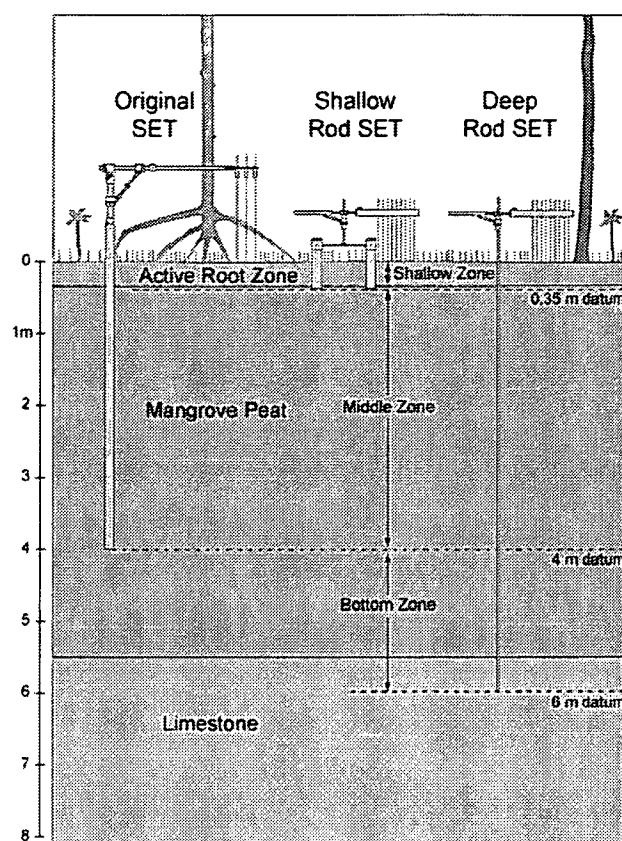


Fig. 1.8. The surface elevation table – marker horizon (SET-MH) method to measure change in elevation of the mangrove sediment surface and vertical sediment accretion, and three different lengths of rods employed in different versions of SET-MH, to calculate shallow subsidence or expansion at different depths in the sediment profile (Cahoon and Hensel, 2006).

Several methods have been employed to observe trends in coastal wetland sedimentation, however, these methods do not account for subsurface factors that influence changes in the elevation of the sediment surface. Soil horizon markers provide accurate measurements of accretion but do not enable measurement of erosion. For example, powdered feldspar clay can be spread over the sediment surface (e.g., Cahoon and Turner, 1989; Krauss et al., 2003). Soil plugs are cut out of the sediment containing the marker horizon and vertical accretion is measured as the distance from the top of the plug to the horizon marker over the time period from when the horizon marker was deposited (e.g., Krauss et al., 2003).

Stakes have been used extensively to observe rates of change in sedimentation and surface elevation of mangroves (Lee and Partridge, 1983; Krauss et al., 2003), river and stream banks (Bradbury et al., 1995), estuaries, mud flats (Kirby et al., 1993), and salt marshes (Lee and Partridge, 1983). Stakes are typically inserted into the sediment at intervals along transects positioned perpendicular to the coastline from low to high water across the coastal system (Lee and Partridge, 1983; Kirby et al., 1993; Bradbury et al., 1995). The initial height of the top of the stakes above the sediment surface, and changes in the height over time, are recorded (Lee and Partridge, 1983; Kirby et al., 1993; Bradbury et al., 1995). In high energy environments such as river and streambanks, mudflats, and estuaries, narrow steel rods 1-2 m long, 10 mm diameter, have been successfully used (Kirby et al., 1993; Bradbury et al., 1995). Sedimentation stakes allow for observations of shallow subsurface processes such as shallow subsidence, belowground productivity from fine root growth, and belowground decomposition of fine root material, down to the depth of the inserted stake, and allow observations of both sediment accretion and erosion. If the stakes are inserted through the soil profile until they reach consolidated basal sediment, stakes could also be used to capture all processes controlling the elevation of the sediment surface.

Measurement of  $^{137}\text{Cs}$  and excess  $^{210}\text{Pb}$  activity in shallow sediment cores may provide an accurate estimate of rates of change in mangrove surface elevation over recent decades, through to the depth of the relevant subsurface soil horizon (e.g., Lynch et al., 1989; Goodbred and Kuehl, 1998; Donnelly and Bertness, 2001), which can then be compared to the relative sea level change rate as measured by the closest tide gauge. However, this is expensive, especially if multiple cores are taken in an attempt to characterize the entire site. This method does not account for subsurface processes that affect the elevation of the mangrove surface that occur below the depth of the cores, and there are several potential sources of error, including that the sediment profile can be disturbed from bioturbation as well as abiotic processes.

In conclusion, to determine trends in mangrove site-specific relative sea-level and to predict responses to regional relative sea-level rise, the current best available technology involves monitoring trends in sediment

surface elevation, such as SET and stake methods inserted through the profile to reach consolidated basal sediment, or similar technology. Observations of disparate trends in sediment elevation within an individual mangrove (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007) highlight the importance of designing monitoring studies to characterize entire sites. Observations from studies using SET-MH technology that subsurface factors exert substantial and in some cases primary control on sediment elevation (e.g., Krauss et al., 2003; Rogers et al., 2005a.b; Whelan et al., 2005; Cahoon et al. 2006) and observations of substantial control of mangrove sediment elevation from groundwater changes primarily from the deepest soil horizon adjacent to bedrock (Whelan et al., 2005) highlight the need to employ sediment elevation monitoring methods that monitor the entire soil profile.

#### **1.5.7. Predicting Mangrove Responses to Projected Trends in Site Specific Relative Sea-Level**

Once information is obtained on mangrove resistance to projected relative sea-level rise, based in part on measurements of trends in sediment elevation, additional information is needed to predict how mangrove position will change as site specific relative sea-level changes. Predicting changes in mangrove position in response to a change in site specific sea-level is one aspect of assessing mangrove resilience to this climate change outcome.

The future position of the mangrove landward margin can be predicted based on the following information: (i) Current boundary delineation; (ii) The mangroves' physiographic setting, which includes the slope of adjacent land and presence of any obstacles (e.g., roads, development, seawalls) to landward mangrove migration through the upper projection for landward transgression of the landward mangrove margin (Section 3.8); (iii) Projections for regional relative sea-level rise (Section 3.1); (iv) Projections for trends in changes in the elevation of the mangrove sediment surface (Section 3.4); and (v) Observations of past trends in the mangrove's position (Section 3.3). A determination for individual mangrove sites if a significant correlation exists between the observed rate of change in position of the mangrove margin and rate of change in relative sea level is needed to determine if there is a basis for using projected rates of change in relative sea level to contribute to

estimating the future change in mangrove position (Section 3.6) (Saintilan and Wilton, 2001; Wilton, 2002). Additionally, information on the contribution of surface vs. subsurface controls on sediment elevation (Section 3.5) can help identify appropriate adaptation options. GIS techniques can then make use of this information to predict how the landward and seaward mangrove margins will change position in response to changes in relative sea-level (Section 3.9).

For sites where remotely sensed imagery dating back several decades is available, it is possible to predict future horizontal movement of mangrove seaward boundaries by reconstructing the historical position and extrapolating the observed movement into the future. Remotely sensed imagery and GIS techniques have been used to assess changes in mangrove and other habitat boundaries (e.g., Woodroffe, 1995; Solomon et al., 1997; El-Raey et al., 1999; Wilton and Saintilan, 2000; Saintilan and Wilton, 2001). For many mangrove sites, it is not possible to identify the landward mangrove margin with any confidence from interpretation of aerial photos and satellite imagery (Woodroffe, 1995; Solomon et al., 1997; El-Raey et al., 1999; Wilton and Saintilan, 2000; Saintilan and Wilton, 2001). In some cases, this is due to the difficulty in differentiating the signatures of mangrove versus upland forest canopy cover. This method accounts for all factors affecting the horizontal position of the mangrove seaward margin, and provides an accurate way to predict future movement. Extrapolations from this method can be adjusted accordingly for sites where changes in relative sea-level are demonstrated to have been a primary control on mangrove position (Section 3.9).

Several studies have assessed coastal vulnerability to sea-level rise by establishing the locations and elevations of coastal habitats and coastal development and use sea level rise projections to estimate what portions of the shoreline will be affected (Woodroffe, 1995; Solomon et al., 1997; El-Ray et al., 2003; South Pacific Applied Geoscience Commission, 2003). The study designs were simplistic by not accounting for trends in sediment elevation of the coastal system. Some studies have employed the Bruun rule to estimate mangrove erosion based on projected sea-level rise, which is likely to result in poor results compared to employing site-specific assessments, because mangroves are not expected to respond in

accordance with Bruun rule assumptions, because mangroves have different sediment budget processes than beaches, and because predictive models of coastal erosion are not suitable for small-scale, site-specific estimates (Bruun, 1988; List et al., 1997; Komar, 1998; Pilkey and Cooper, 2004).

## **1.6. THESIS AIMS AND HYPOTHESES**

Two primary aims of this thesis are to:

- (i) Identify and implement a method, suitable for broad application in the Pacific Islands region, to accurately assess site-specific mangrove vulnerability and predict responses to projected relative sea level rise. The comprehensive method needs to assess both site-specific resistance, used in this study to mean: will the mangrove keep pace with rising sea-level, such that sea-level is not rising relative to the mangrove sediment surface, and as a result there is no alteration to the mangrove's functions, processes and structure (Odum, 1989; Bennett et al., 2005); and resilience, used in this study to mean: what is the capacity of the mangrove to naturally migrate landward in response to a rise in sea-level relative to the mangrove surface, and will the mangrove absorb and reorganize from the stress of rising sea-level to maintain its functions, processes and structure (Carpenter et al., 2001; Nystrom and Folke, 2001); and
- (ii) Identify priority technical and institutional capacity-building needs of Pacific Island countries and territories to assess mangrove vulnerability to change in sea-level, how mangroves will respond to projected changes in sea-level, and to adapt to predicted mangrove responses.

The aims of the research as stated here are based on the reviews conducted in this introductory chapter, which identified gaps in knowledge for assessing mangrove vulnerability and predicting responses to relative sea-level rise. This is the first study of its kind to assess aspects of both site-based resistance and resilience to projected trends in regional relative sea-level. The extensive literature of paleoenvironmental shoreline reconstructions (e.g., Woodroffe et al., 1985; Sugimura et al., 1988; Ellison and Stoddart, 1991; Woodroffe, 1992, 1995; Shaw and Ceman, 1999; Berdin et al., 2003; Ellison

1993, 2000, 2006) do not provide information needed to predict future changes over human time scales, and effects of modern anthropogenic stressors may alter mangrove systems sufficiently causing substantial deviation from paleoenvironmental responses to change in sea-level. Observations of mangrove sedimentation rates (e.g., Thom, 1984; Ellison and Stoddart, 1991; Parkinson et al., 1994; Woodroffe, 1987, 1995, 2002) alone provide an unreliable indicator of vulnerability to sea-level rise as sedimentation is but one of several controls on mangrove sediment elevation (Krauss et al., 2003; Rogers et al., 2005a.b; Whelan et al., 2005; Cahoon et al. 2006). Recent studies employing the SET method (Krauss et al., 2003; Rogers et al., 2005a.b; Whelan et al., 2005; Cahoon et al. 2006) accurately assess vulnerability to change in sea-level by monitoring trends in sediment surface elevation through the entire soil profile. However, these studies have not included adequate spatial sampling to address spatial variability in trends in elevation within the site (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007), and for sites observed not to be keeping pace, there have been no assessments of resilience. Other assessments, using interpretation of remotely sensed imagery and using a GIS, have employed predictive models not suited for mangroves or site-specific assessments, and have not accounted for trends in coastal ecosystem sediment elevation when assessing vulnerability and response to change in sea-level (Gilman, 1990; Woodroffe, 1995; Solomon et al., 1997; El-Raey et al., 1999; Wilton and Saintilan, 2000; Saintilan and Wilton, 2001; El-Ray et al., 2003). Given the poor state of understanding of the relationship between geomorphic setting and change in sediment surface elevation, and because there may be many different geomorphic settings within a site, until reliable predictive elevation models are developed for mangrove ecosystems, site-specific monitoring is necessary to assess vulnerability and predict responses to sea-level rise.

There has been no previous assessment of the Pacific Island region's capacity to determine mangrove vulnerability and predict responses to changes in sea-level, and preparedness for adaptation (Section 6.2) (Nurse et al., 2001; Mimura et al., 2007). Previous studies of preparedness for vulnerability assessments and adaptation, synthesized by IPCC for small island states (Nurse et al., 2001; Mimura et al., 2007) and reported by individual countries in their National Communication reports to UNFCCC

(Federated States of Micronesia, 1997; Government of Samoa, 1999; Government of Tuvalu, 1999; Kiribati Government, 1999; Republic of Nauru, 1999; Republic of Vanuatu, 1999; Papua New Guinea Government, 2000; Republic of the Marshall Islands Environmental Protection Authority, 2000; Republic of Palau Office of Environmental Response and Coordination, 2002; Solomon Islands Government, 2004; Kingdom of Tonga, 2005) reveal this information gap. Instead, past studies have highlighted in general terms the vulnerability of island ecosystems and resources, development and people, and identified adaptation options, with a focus on adjusting to sea-level rise and storm surges (Mimura et al., 2007).

Hypotheses related to the first aim are:

- (i) Mangroves in an estuarine geomorphic setting are more resistant to regional relative sea-level rise than those found in an embayment setting. This is hypothesized to be the case because estuarine mangroves are expected to have a higher rate of change in the sediment surface elevation due to having greater hydrologic turnover, more freshwater inputs and lower salinity, higher primary productivity, and higher sedimentation rates than embayment mangroves (Thom, 1982, 1984; Twiley, 1988; Woodroffe, 1987, 2002);
- (ii) Mangroves located in relatively undisturbed catchments demonstrate a significant positive correlation between the rate of movement of mangrove margins and rate of change in regional relative sea-level; and
- (iii) Mangrove sedimentation rates are not equal to rates of change in sediment elevation.

The first aim is implemented through the analysis of five categories of information:

- (i) Trends in mangrove sediment elevation, accounting for all controls on elevation through the sediment profile;
- (ii) Trends in regional relative sea-level;
- (iii) Projections for site specific relative sea-level change, based on the previous two components combined with locally-applied projections for change in global sea-level;
- (iv) Past trends in movement of the mangrove seaward margin, and a determination of whether or not change in site-specific relative sea-level



can be inferred to be a primary factor controlling mangrove position (based on an observation of a significant positive correlation between the rate of movement of mangrove margins and rate of change in regional relative sea-level); and

- (v) The mangrove's physiographic setting (slope of the land adjacent to the mangrove, slope within the mangrove, and presence of obstacles to landward migration).

Hypotheses related to the second aim are:

- (i) Pacific Island countries exhibit substantial gaps in capacity to assess mangrove resistance and resilience to changes in regional relative sea-level; and
- (ii) There is a low degree of capacity to implement methods for adaptation to mangrove responses to climate change in the Pacific Islands region.

The second aim is implemented by assessing the technical and institutional capacity of Pacific Island Countries and territories with indigenous mangroves to predict site-specific mangrove responses to projected changes in regional relative sea-level, and preparedness for adaptation to mangrove ecosystem responses to changes in sea-level.

## **1.7. THESIS STRUCTURE**

This introductory chapter has described threats to mangroves from climate change. A review of mangrove ecosystem values provided the basis for the importance of this coastal ecosystem and the need to understand the threat posed by changes in relative sea-level and other climate change outcomes. A review of the state of knowledge of mangrove responses to climate change outcomes other than change in sea-level, ranging from changes in the frequency and levels of extreme high water events to human responses to climate change, was provided. Influences on global sea-level over glacial and human time scales were identified, and a review of Holocene sea-level patterns and causes in the Pacific Islands region and the Samoa archipelago were reviewed. A review was conducted of coastal wetland responses to change in relative sea-level based on paleoenvironmental reconstructions

and recent observations. The numerous surface and subsurface and feedback mechanism controls on mangrove sediment elevation, influences of the geomorphic and physiographic setting, species composition, and cumulative effects of stressors on mangrove resistance and resilience to changes in sea-level were highlighted as factors contributing to site-specific mangrove responses to change in regional relative sea-level. A review of methods to measure trends in site-specific relative sea-level in mangroves was conducted, highlighting the importance of a sampling design that adequately characterizes a site and the need to employ sediment elevation monitoring methods that account for subsurface processes throughout the entire soil profile to bedrock. The final two sections of the Introduction explain the thesis aims, hypotheses and structure.

Chapter 2 describes the study area, including the status, trends, distribution and biodiversity of mangroves of the Pacific Islands region, and characteristics of the three mangrove study sites in American Samoa.

Chapter 3 describes the methods employed to predict three American Samoa mangroves study sites' vulnerability and responses to projected regional relative sea-level rise, while Chapters 4 and 5 present the results and discussion from these study components, respectively. Study components include

- (i) Projections for relative sea-level rise through the year 2100;
- (ii) Observing trends in frequency and elevation of extreme high water events and causes;
- (iii) Historical reconstruction of mangrove margin position from analysis of a time series of remotely sensed imagery and a geographic information system;
- (iv) Observations of trends in changes in sediment elevation over a period of years through the use of stakes;
- (v) Observations of sediment accretion rates over decades from  $^{137}\text{Cs}$  and Excess  $^{210}\text{Pb}$  activity dating;
- (vi) Assessment of the importance of change in regional relative sea-level as a control on observed mangrove margin movement, through determination of a significant positive correlation between the observed change in regional relative sea-level and observed rate of movement of the mangrove margin;

- (vii) Observations of past and predicted future trends in site-specific relative sea-level, to determine mangrove resistance to regional relative sea-level rise, through comparison of the observed rate of change in mangrove sediment elevation, observed regional relative-sea level rise rate, and projected future scenarios of regional relative sea-level rise rates;
- (viii) Assessment of mangrove physiographic settings to determine resilience to any predicted landward migration in response to site-specific relative sea-level rise;
- (ix) Predicted change in mangrove area and position by 2100; and
- (x) Regional extrapolation from American Samoa predicted mangrove responses to regional relative sea-level rise.

Chapter 6 presents the results of an assessment of the capacity of the 16 Pacific Island Countries and territories with indigenous mangroves to assess mangrove vulnerability to climate change and adapt to mangrove responses to these factors. Requisite technical and institutional capabilities are based on the methods employed in American Samoa. Results are used to identify regional capacity-building priorities. Priorities include establishing a regional mangrove monitoring network, and augmenting the efficacy of management frameworks to avoid and minimize adverse effects on mangroves and other valuable coastal ecosystems and plan for any predicted landward mangrove migration.

Chapter 7 identifies actions that management authorities and local communities can employ to adapt to predicted mangrove responses to climate change effects, including projected regional relative sea level rise. Principles and approaches for site planning and community-based coastal management are described. Alternative adaptation methods are identified, including: “no regrets” reduction in anthropogenic stresses, establishment and management of protected areas that include mangroves, mangrove rehabilitation, monitoring and outreach.

Chapter 8 presents conclusions on the results from American Samoa and regional capacity assessment. Results are compared to the reviewed literature to document how the state of knowledge for predicting mangrove

responses to projected relative sea-level rise has been advanced. Finally, it is explained how results fulfill the thesis aims and hypotheses.

Appendix 1 describes a mangrove restoration project conducted in American Samoa, which employed community-based and low-cost methods. A preliminary assessment of the performance of the restoration project is presented. Appendix 2 contains metadata for historical aerial photos and space imaging of American Samoa mangrove study sites. Appendix 3 contains the questionnaire used for the regional capacity survey.

## **Chapter 2**

### **Study Area**

#### **2.1. PACIFIC ISLAND MANGROVES**

##### **2.1.1. Status, Trends and Documented Losses**

The cumulative effects of natural and anthropogenic pressures make mangrove wetlands one of the most threatened natural communities worldwide. Roughly 50% of the global area has been lost since 1900 due primarily to human activities such as conversion for aquaculture and filling (IUCN, 1989; Ramsar Secretariat, 1999; Valiela et al., 2001). The global average annual rate of mangrove loss is 1 to 2%, exceeding the rate of loss of tropical rainforests (0.8%) (Valiela et al., 2001; FAO, 2003; Wells et al., 2006; Duke et al., 2007). While the validity of these figures, based on data from the Food and Agriculture Organization of the United Nations (FAO, 2003) are questionable, losses during the last quarter century range between 35 and 86% (Primavera, 1997; Smith et al., 2001; FAO, 2003; Duke et al., 2007).

Islands are one of the more vulnerable coastal types to sea-level rise. The estimated area of mangroves in the Pacific Islands is 524,369 ha with largest areas in Papua New Guinea (372,770 ha), Solomon Islands (64,200 ha), Fiji (41,000 ha), and New Caledonia (20,250 ha) (Ellison, 2000). The reduction in mangrove area from 1980 to 2000 was roughly estimated to be 18%, representing a loss of over 124,000 ha, in the 17 Pacific Island countries and territories reported to have indigenous mangrove ecosystems (American Samoa, Fiji, Guam, Kiribati, Marshall Islands, Micronesia, Nauru, New Caledonia, Niue, Northern Mariana Islands, Palau, Papua New Guinea, Samoa, Solomon Islands, Tonga, Tuvalu, Vanuatu) (FAO, 2006). Threats to mangroves include filling; nutrient, freshwater, and pollutant inputs; excessive sedimentation; harvesting and clearing mangrove vegetation; changing sediment budgets such as from the construction of seawalls and alterations within the wetland's contributing watershed area; displacing native species

with alien invasive species; and harming vegetation from insect infestations, fungal flora pathogens, and other diseases (Scott, 1993; Ellison, 1999; Donnelly and Bertness, 2001; Saintilan and Wilton, 2001).

There are roughly 17 million ha of mangroves worldwide (Valiela et al., 2001; FAO, 2003). The Pacific Islands contain roughly 3% of global mangrove area, a small area in global terms, but each island group has a unique mangrove community structure (Ellison, 2000) and mangroves provide site-specific functions and values, for instance, where reducing coastal erosion and protection from storms are site-specific mangrove ecosystem services (Section 1.2) (e.g., Gilman, 1998; Lewis, 1992). Also, portions of a mangrove species' range, on remote islands, may be genetically isolated, resulting in unique varietal characteristics (Duke, 1992; Ellison, 2004). Pacific Island governments have recognized the value of mangroves and the need to augment conservation efforts (e.g. South Pacific Regional Environment Programme, 1999a). There is little available quantitative information on trends in area or health of Pacific Island mangroves due to limited monitoring, and many of the area estimates presented in this section are based on outdated primary sources.

In Papua New Guinea there are between 353,770 – 391,770 ha of mangrove (Scott, 1993). Southern Papua New Guinea mangroves have the highest global mangrove diversity with 33 mangrove species and 2 mangrove hybrids, located at the center of the Indo-Malayan mangrove center of diversity (Ellison, 2000; Duke et al., 1998; Duke, 2006). Most mangrove areas are on the south coast, where major mangrove areas include the Fly delta of about 87,000 ha (Robertson et al., 1991) and the Purari Delta complex of about 134,000 ha (Scott, 1993). Mangrove areas adjacent to population centers have been documented to have experienced degradation, such as from clearing for shanty town development (Ellison, 1999).

The Solomon Islands contain about 64,200 ha of mangroves (Hansell and Wall, 1976). Main mangrove areas are found on Isabel, New Georgia and Malaita Islands. There is no documentation of mangrove losses; potential future threats are from logging, mining and filling for development (Scott, 1993).

There are an estimated 41,000 ha of mangrove in Fiji (Watling, 1985). More than 90% of mangroves are on Viti Levu and Vanua Levu (Watling, 1985). The largest areas are on the Ba, Rewa and Nadir Rivers on Viti Levu and the Labasa River on Vanua Levu. The largest mangrove area loss of 2,334 ha was in the Labasa Delta from poldering for agriculture by the Colonial Sugar Refining Company (Lal, 1991). An estimated 6% of the mangrove area (2,457 of 41,000 ha) was lost from conversion to other uses (Ellison, 1999). In the 1970s, 308 ha were cleared for development of an aquaculture facility.

New Caledonia has about 20,250 ha of mangrove (Maragos, 1994; Metz, 2000). Largest areas are on the west coast of Grande Terre. Losses of about 380 ha were documented from urban development and garbage dumps within mangrove areas located around Noumea (Scott, 1993; Ellison, 1999).

The Federated States of Micronesia has about 8,564 ha of mangrove (MacLean et al., 1998, compilation of previous assessments interpreting aerial photography from 1976 combined with 1983 fieldwork). Scott (1993) reports that 65% of mangrove area occurs on the island of Pohnpei, 18% of Kosrae and 14% on Yap.

Palau contains 4,500 ha of mangrove (Maragos, 1994; Metz, 2000). Most (> 85%) of the mangroves occur on Babeldao Island primarily in estuaries, while mangroves also occur around Koror, Peleliu and some of the smaller offshore islands.

There are between 2,500 – 3,000 ha of mangrove in Vanuatu (Scott, 1993). The largest mangrove areas are two stands on Malakuka, the Port Stanley/Crab Bay area and the Maskelyne Islands/Lamap area (Scott, 1993). Losses were documented in Port Vila from development (Minagawa, 1992).

Tonga contains 1,305 ha of mangrove (Wiser et al., 1999). Mangrove areas were recently lost from filling at Popua and Sopa, and the government has authorized the clearance of all other large mangrove areas (Fifita, 1992; Ellison, 1999).

Samoa has an estimated 700 ha of mangrove (Pearsall and Whistler, 1991). Schuster (1992) documents the loss of 0.65 ha of mangrove due to the construction of an aquaculture facility in 1978.

There are an estimated 258 ha of mangrove in Kiribati (Nenenteiti Teariki-Ruatu, personal communication, February 2005, Republic of Kiribati Ministry of Environment, Lands, and Agricultural Development).

There are an estimated 70 ha of mangrove on Guam (Scott, 1993). Filling for development around Apra Harbor caused extensive mangrove losses, while filling for urban development and aquaculture facilities are more recent causes of losses (Scott, 1993).

There are about 40 ha of mangrove in Tuvalu (Scott, 1993). Small mangrove stands occur on five of the nine islands in the Tuvalu group; Mangroves occur on three of the limestone islands (Vaitupu, Nanumanga and Niutao) and two of the atolls (Funafuti and Nui). There is no documentation of mangrove losses (Scott, 1993).

There are about 4 ha of mangrove in the Marshall Islands (John Bungitak, personal communication, August 2005, Environmental Protection Agency, Republic of the Marshall Islands). Small mangrove stands are known to occur on Jalwoj, Arno and Aelonlaplap atolls in the southern part of the archipelago, with small mangroves found in inland depressions on islets in the northern portion of the archipelago (Scott, 1993).

The Northern Mariana Islands contain about 5 ha of mangrove (Gilman, 1998, 1999b). A large proportion of the Northern Mariana Islands' mangroves were lost during the 30-year period (1914-1944) of Japanese occupation when much of the land was cleared for cultivation of sugarcane and coconut, and for fortification, construction of airstrips, and construction of barracks prior to World War II (Coastal resources Management Office, 1991). The U.S. military built additional airstrips and barracks after taking control of Saipan in 1944 (Coastal Resources Management Office 1991, Soil Conservation Service 1989). During and immediately after the war, large areas of land that had been used for agriculture by the Japanese, likely including cultivated wetlands on Saipan's western coastal plain, were bulldozed and covered with crushed limestone and in some places paved by the American military for use as base facilities (Butler and DeFant 1991). More recently, incremental losses from illegal and permitted filling for roads, buildings and agriculture are occurring (Gilman, 1998, 1999b).



On Nauru, a single stand of mangrove of about 1-2 ha is located on the island's northeast coast (Scott, 1993).

A Niue government focal point reported that there are no mangrove wetlands in Niue (personal communication, 10 June 2005, Fiafia Rex, Fisheries Division, Niue Department of Agriculture Forestry & Fisheries). This is consistent with Scott (1993) who reports no occurrence of mangroves in Niue's coastal zone. While one true mangrove species *Excoecaria agallocha* is documented to be present in Niue (Yuncker, 1943; Ellison, 1999), in Niue, this species is only found in dry littoral forest.

### **2.1.2. Geomorphic Settings, Distribution and Biodiversity**

Mangrove wetlands are found in the intertidal zone, between mean sea level and high tide elevation, of tropical and subtropical coastal rivers, estuaries and bays. In general, mangroves are most extensive on macro-tidal coastlines or on low gradient coasts, in areas with a large supply of fine-grained sediment, particularly in large embayments or deltas with strong tidal currents but low wave energy, and are most productive in areas with high rainfall or relatively large freshwater supply from runoff or river discharge (Woodroffe, 1992, 2002). However, mangroves also occur in areas with small tidal range where there is sufficient fine sediment input. In the Pacific Island region, mangroves primarily occur in four geomorphic settings:

- (i) Deltas/estuaries found on 'high' islands with well-developed river systems (river-dominated allochthonous, Thom [1982, 1984]);
- (ii) Embayments/lagoons/harbors (drowned bedrock valley, Thom [1982, 1984]);
- (iii) Reef flats (carbonate sand/shingle barrier and Quaternary reef top, Thom [1982, 1984]); and
- (iv) Inland, depressions usually found on small, 'low' islands (basin, Cintron et al. [1985]) (Woodroffe, 1987).

Sedimentation rates tend to range from highest to lowest in this order. There are few tide-dominated mangroves, as most of the Pacific Island mangroves exist in areas with microtidal regimes of around 1 m. And there are few wave-dominated mangroves, as areas where wave energy is high tend to

have extensive reef development protecting landward mangroves (Woodroffe, 1987).

Mangroves can tolerate a wide variation in water and soil salinity (e.g., Cintron et al., 1985) and occur in reduced and acidic soil conditions (e.g., Chapman, 1976). While saltwater is not necessary for the survival of any mangrove species, this provides mangroves with a competitive advantage over salt-intolerant vascular plant species that lack adaptations for survival in the intertidal zone, including long hydroperiod and anaerobic soil conditions of mangrove habitats (Davis, 1940; Lugo, 1998). Dominant plant species in mangrove ecosystems possess several adaptations for existence in the intertidal zone, in unstable, saline and anaerobic soils, including pneumatophores and prop roots, aboveground roots, which transfer oxygen to belowground roots and provide structural support; salt exclusion through membranes in cells at the root surface; removal of salt through special glands in leaves in some species and concentrating salt and shedding these leaves in other species; minimizing freshwater loss through evapotranspiration by restricting the stomata diameter and altering the orientation of their leaves to reduce leaf surface exposure to direct sunlight; and in some species, buoyant and viviparous seeds (germinate while attached to the parent tree) (Chapman, 1976; Mitsch and Gosselink, 1993).

Fig. 2.1 shows the distribution of mangrove species in the Pacific Islands region, constituting a total of 31 true mangrove species and 5 hybrids (Ellison, 1995, In Press b). Except for *R. mangle*, mangroves found in the Pacific Islands region are of the Indo-Malayan assemblage (Ellison, 1995, In Press b). There is a decline in Pacific Island mangrove diversity from west to east across the Pacific, and indigenous mangroves of the Indo-Malayan assemblage reach an eastern limit at American Samoa where there is an estimated 52 ha of mangroves remaining with three mangrove species (Ellison, 1995, 1999, In Press b). In the Pacific Islands region, mangroves do not naturally occur further east of American Samoa due to difficulty of propagule dispersal over such a large distance and historic loss of habitat during Holocene sea level changes (Ellison and Stoddart, 1991). In addition, some islands may have lower numbers of mangrove species due to a lack of suitable intertidal habitat (Ellison, 2001). Mangroves are recent human

introductions in Hawaii, USA and French Polynesia (Allen, 1998; Ellison, 1999; Smith, 2005; Chimner et al., 2006).

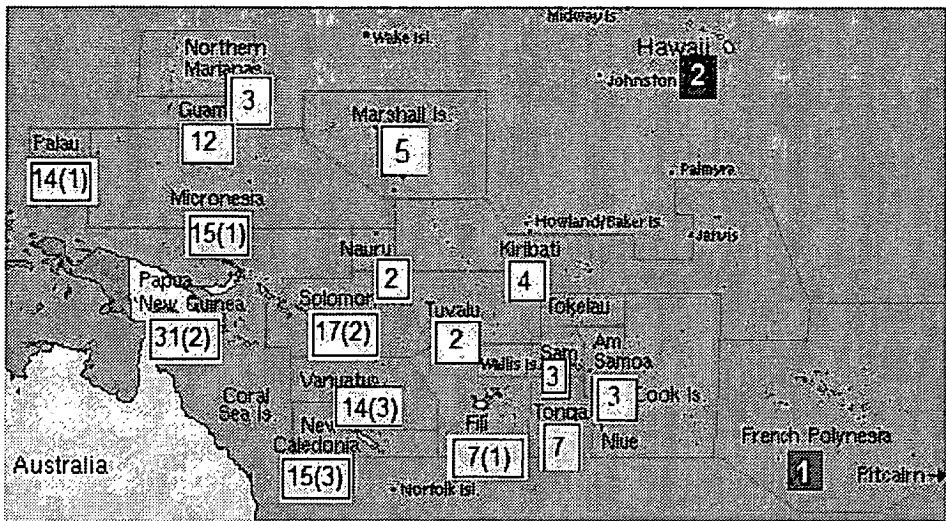


Fig. 2.1. Mangrove species distributions in the Pacific Islands region (Ellison, 1999, In Press b). Yellow squares give the number of mangrove species in the 16 countries and territories where mangroves are indigenous, blue squares are the two locations where mangroves are human-introductions. The number of mangrove hybrid species is in parentheses.

### 2.2. AMERICAN SAMOA

American Samoa is the eastern portion of the Samoan archipelago, is a territory of the United States, located in the central western Pacific between 168° and 173° W longitude and 13° and 15° S latitude (Fig. 2.2). American Samoa has a total land area of about 197 km<sup>2</sup>. The island of Tutuila comprises 71% of this total (U.S. Department of the Interior, 1997). About 95% of American Samoa's total estimated population of 63,100 lives on the island of Tutuila (U.S. Department of the Interior, 1997). The Samoa archipelago was settled 3,000 – 2,800 years BP (White and Allen, 1980; Kirch, 1997). Most of American Samoa's commercial and residential development is located around the perimeter of Pago Pago Harbor and on the Tafuna coastal plain in the central part of Tutuila. Only 26% of Tutuila Island is developable (land with ≤ 8 degree maximum slope) due to steep

inland mountains. The scarcity of flat, developable land on Tutuila has resulted in the concentration of development in narrow strips of coastline flanked by narrow, steep ridges. Population is projected to approximately double from 57,291 in 2000 to 107,386 by 2025 (American Samoa Department of Commerce, 2000).

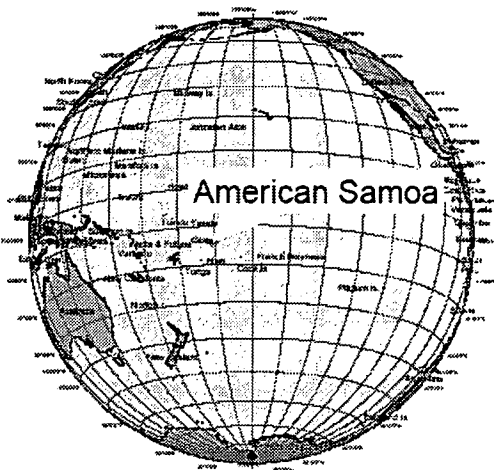


Fig. 2.2. Location of American Samoa, the eastern portion of the Samoan archipelago.

There has been disagreement over the origin of the Samoan archipelago, some arguing that the islands were formed from rifting or crustal plate activity associated with the northern terminus of the Tonga Trench, supported by the presence of numerous young volcanism on what should be the oldest western island of Savai'i (Amerson et al., 1982; U.S. Soil Conservation Service, 1984), others arguing that the islands follow a classic hot spot track (U.S. Department of the Interior, 1997). Recent observations of an active underwater volcano (Vailulu'u) located far to the east of the Tonga Trench, as well as geodetic reconstructions which found that the northern terminus of the Tonga arc was too far west of Samoa until 1-2 my ago to have caused volcanism, support the formation of the chain through hot-spot volcanism, with an east-to-west age progression (Hart et al., 2000, 2004). The oldest island is Savai'i, dated to have formed 5 million years before present (Hart et al., 2004). Tutuila has been dated at 1.26 million years (U.S. Department of the Interior, 1997).

Tutuila was created through a series of volcanic eruptions, including renewed volcanism, which began in the late Pleistocene less than 1 million years ago, which continues to the present. Subsequent stream erosion and subsidence have resulted in the current steep ridges, narrow stream valleys and deeply embayed coastline, which is characteristic of cycles of erosion and submergence. Soils of mountain areas are shallow to deep and well drained. Over half of the area of Tutuila has slopes of 70 percent or more (U.S. Soil Conservation Service, 1984). The relatively flat coastal plain located between Nu'uuli and Leone villages, the only sizeable area with gentle slopes, formed through recent lava and tuff laid upon submerged reef (Amerson et al., 1982). Nu'uuli lagoon, drained by a large canyon, was formed by shoaling of water by a recent lava flow and formation of the Coconut Point sand spit.

Tutuila is 32 km long, 2-9 km wide, with steep terrain descending from a central ridge, which reaches 524 m in elevation, with an average elevation of 300 m (American Samoa Coastal Management Program, 1992). The current mixed semidiurnal tidal range, based on Pago Pago records, is about 1.1 m. Mean annual rainfall is 312 to 563 cm, with a rainy season from November to March and dry season from June to September. Mean annual temperature is 26.7° C, with an annual range between 21 and 32° C (U.S. Soil Conservation Service, 1984; Scott, 1993). Tropical depressions and occasional cyclones can occur during the austral summer, usually between December and March; tropical storms of sufficient strength to cause damage to mangroves occur every 4-5 years. Typical weather and climate conditions occurred during the study period, with no damaging storms.

### **2.3. AMERICAN SAMOA MANGROVES**

There are nine mangrove wetlands in American Samoa, located on Tutuila and Aunu'u islands, with an estimated combined area of 52.3 ha (Fig. 2.3) (Bardi and Mann, 2004). While mangroves were once prominent features at the mouths of most freshwater streams in American Samoa, the majority of mangrove area has been filled since the early 1900's (Amerson et al., 1982; American Samoa Coastal Management Program 1992; Bardi and Mann,

2004). At least 10 ha of mangroves have been lost from Pago Pago Harbor due to harbor development (Scott, 1993). Filling for development in 1991 caused the loss of 7.8 ha of Leone mangrove and 25 ha of Nu'uuli mangrove (Ellison, 1999).

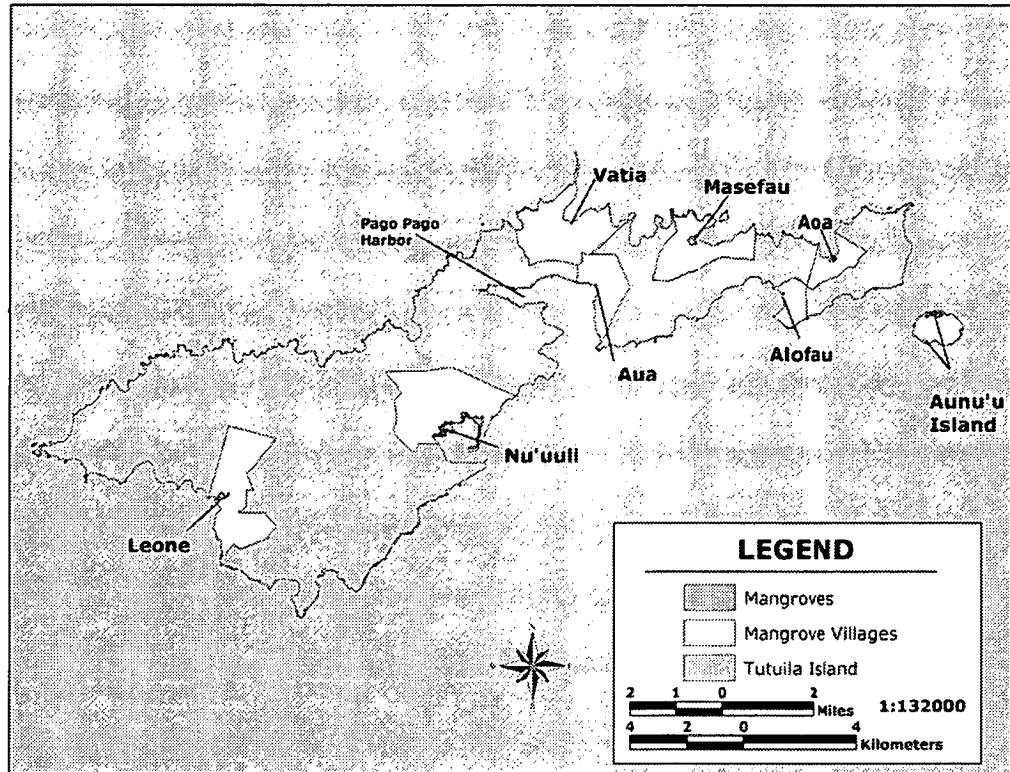


Fig. 2.3. Location of American Samoa's nine mangroves on Tutuila and Aunu'u Islands, including the three study sites, Leone, Nu'uuli, and Masefau mangroves.

Three mangrove species and several mangrove associate species are present in American Samoa's mangrove communities (Amerson et al., 1982; Bardi and Mann, 2004). American Samoa mangroves are dominated by a single tree species, *Bruguiera gymnorhiza* (L.) Lamx. (oriental mangrove), with *Rhizophora mangle* L. (red mangrove) found primarily along mangrove seaward margins, along tidal creeks, and with a few individual trees interspersed in areas dominated by *B. gymnorhiza* (Amerson et al., 1982; Scott, 1993; Bardi and Mann, 2004). *Xylocarpus granatum* (monkey puzzle-nut or cannonball tree) is rare, with only a few individual trees found at Nu'uuli and Aunu'u mangroves (Amerson et al., 1982; Bardi and Mann,

2004). The species of *Xylocarpus* found in American Samoa was previously believed to be *X. moluccensis*, but recently has been determined to be *X. granatum* (Duke, 2006). *R. mangle samoensis* is found in New Caledonia, Fiji, Tonga, Samoa and American Samoa, believed by some authors to be a distinct species (Woodroffe, 1987), however others believe it is the same species as *R. mangle* (Ellison, 1991, In Press b). *R. mangle* is the only mangrove species found in both the Indo-Malayan and American mangrove assemblages (Tomlinson, 1986). In American Samoa, due to the absence of landward zone mangrove species, such as *Ceriops*, *B. gymnorhiza* occupies most of the mangrove area and allows for the formation of marshes landward of some of the mangrove areas (American Samoa Coastal Management Program, 1992; Ellison, 1999). Table 2.1 describes the mangrove-associated flora of American Samoa mangroves.

Table 2.1. American Samoa mangrove-associated flora (Amerson et al., 1982; American Samoa Coastal Management Program 1992; Scott, 1993; Ellison and Oxley, 1998).

Scientific Name	Samoan and English Names	Identification
<i>Acrostichum aureum</i>	Samoan: Sa'ato, ulu sa'ato	Robust fern, 1-3 m pinnate fronds. Invades following disturbance.
<i>Asplenium nidus</i>		Epiphyte.
<i>Barringtonia racemosa</i> , <i>B. samoensis</i>	Samoan: Falaga	Small to medium sized tree. Typical at landward edge of mangrove.
<i>Clerodendrum inerme</i>	Samoan: Aloalo tai	Sprawling shrub or woody climber. Grows at the edge of the mangrove.
<i>Cyperus alternifolius</i> subsp. <i>flabelliformis</i>	English: Umbrella plant	Sedge. Found commonly in freshwater marshes.
<i>Cyperus javanacus</i>	Samoan: Selesele, nea nana'ai; English: marsh Cyperus	Sedge with slightly 3-angled stems, 40-100 cm tall. Found in saltwater marshes.
<i>Davallia solida</i>		Fine bladed fern.

<i>Derris trifoliata</i>	Samoan: fue'o'ona	Woody vine. Blankets the ground and surrounding vegetation. Odd pinnate leaves with 3 to 7 shiny leaflets. Found at the mangrove fringe and coastal strand, and sometimes at the mangrove landward edge.
<i>Fimbristylis cymosa</i>		Sedge grass with erect 3-angled stems 10-50 cm tall. Found in salt marshes.
<i>Geniostoma insulare</i>	Samoan: Fau	Large tree.
<i>Hibiscus tiliaceus</i>	Samoan: Fau	Tree grows in thickets with tangled branches, up to 15 m tall. Alternate heart-shaped leaves, dark green above and white below. Grows in dense thickets at the mangrove landward margin.
<i>Hoya australis</i>	Samoan: Fue selala	Fleshy leaved vine.
<i>Inocarpus fagifera</i>	English: Tahitian chestnut	Large tree.
<i>Ipomoea pes-caprae</i>	Samoan: Fue moa; English: beach morning glory	Glabrous creeping, non-twining vine, usually restricted to the coastal strand and lowland marshes.
<i>Mariscus javanicus</i>	Samoan: Selesele	Sedge.
<i>Pandanus tectorius</i>	Samoan: Fala	Tree.
<i>Paspalum distichum</i>	English: Knottgrass, saltgrass, knottweed	Creeping wiry grass. Forms meadows in saltwater coastal marshes and invades sand flats and mud flats where mangroves have been cleared.
<i>Pittosporum arborescens</i>		Large tree.
<i>Polypodium scolopendria</i>		Solid bladed fern.
<i>Sesuvium portulacastrum</i>	English: Sea purslane	Succulent, herbaceous halophyte. Found in coastal wetlands, mud flats, and pitted sandstone.
<i>Stenochlaena palustris</i>		Epiphyte.
<i>Taeniophyllum fasciola</i>		Epiphyte. Grows on bark.



<i>Thespesia populnea</i>	Samoa: Milo	Shrub or small tree to 4 m tall, opposite elliptic to obovate leaves, gray-green with white silky hairs on leaves and young branches. Found in the coastal strand and on coralline substrate, and sometimes at the mangrove interface exposed to tidal influence.
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A total of 25 taxa of mangrove-associated algae have been recorded from Upolu Island, Samoa: Mangrove communities dominated by *B. gymnorhiza* contained bostrychietum algae (occurring as epiphytes on mangrove stems and roots or growing on other substrata including basaltic rocks and garbage) (Skelton and South, 2002). Communities dominated by *R. mangle* had two algal assemblages, both which lacked *Bostrychia* species: one assemblage contained a mixture of *Caloglossa* spp. and *Murrayella pericladus*, and was found on pneumatophores, and the second assemblage contained primarily *Polysiphonia*, *Murrayella* and *Cladophora* species growing as a thick mat on rocks and other hard substrata (Skelton and South, 2002).

The predominant soil type within American Samoa mangroves is Ngerungor Variant organic peat, a mixture of peat and basaltic and calcareous sand, comprised of 10-30% organic matter (U.S. Soil Conservation Service, 1984; Ellison, 2001). The color for moist soil from 0-10 cm is a very dark grayish brown (10YR 3/2) organic peat. The color for moist soil from 10-52.5 cm is a very dark brown (10YR2/2) peat (U.S. Soil Conservation Service, 1984). Radiocarbon dates of mangrove stratigraphy from five mangrove sites on Tutuila (Leone, Nu'uuli, Alofau, Aoa, and Masefau) documented sediment accretion rates of between 1.2 – 4.5 mm a<sup>-1</sup>, where radiocarbon dates of the deepest, oldest horizons of the cores ranged between 270 – 3080 BP (Ellison, 2001). Relatively high rates of sediment accretion in two of the mangroves (Masefau, 3.5 mm a<sup>-1</sup> from 410 BP, and Alofau, 4.5 mm a<sup>-1</sup> from 270 BP) likely are the result of the steep gradients on a young volcanic island, and for mangroves within the more densely populated catchments, human land uses have caused increased erosion, with concomitant increased sediment accretion rates in mangroves. The low

organic content of the Tutuila mangrove muds is consistent with there being high terrestrial sediment inputs.

In the three study sites, Nu'uuli, Masefau and Leone mangroves, in typical, undisturbed, mature *B. gymnorrhiza*-dominated communities, tree density is about 1 tree per 4 m<sup>2</sup>. Tree species composition is about 85% *B. gymnorrhiza* and 15% *R. mangle*, and canopy cover is > 85%. Tree height is an average of about 10 m. Average *B. gymnorrhiza* tree diameter at breast height (1.3 m high) is about 14 cm. There are numerous seedlings, about 7.5 seedlings per 1 m<sup>2</sup>, in similar proportions as tree species composition. There are few saplings, about 1 per 25 m<sup>2</sup>. The width of *B. gymnorrhiza*-dominated communities of Nu'uuli mangrove range from < 5 to 150 m from landward to seaward margin. In Masefau and Leone mangroves the *B. gymnorrhiza* communities are about 400 m and 175 m wide, respectively.

In typical, undisturbed, mature *R. mangle*-dominated communities of the three study sites, the width is very narrow,  $\leq 10$  m, located as a fringe seaward of the *B. gymnorrhiza*-dominated community. Tree density is about 1 tree per 2 m<sup>2</sup>. Tree species composition is close to 100% *R. mangle*, and canopy cover is > 85%. Tree height is an average of about 3 m. Average tree diameter at breast height is about 3.5 cm. There are about 4 seedlings per 1 m<sup>2</sup>, almost all *R. mangle*. There are very few saplings, < 1 per 25 m<sup>2</sup>.

The three study sites include the three largest mangrove areas of American Samoa, located on the main island of Tutuila. The largest of these is Nu'uuli (30.69 ha) (Bardi and Mann, 2004), with an approximate center at 170° 42.766' W, 14° 18.844' S. Using the geomorphic classification scheme of Woodroffe (2002) (Section 1.5.1), Nu'uuli best fits into the embayment class (of the many classification schemes available, the one developed by Woodroffe (2002) is used because of its wide applicability and its incorporation of numerous previous classifications). The depth of peat and organic mud ranges from 0.01 m (mangrove trees growing on exposed volcanic rock with pockets of organic mud and peat) to > 1.5 m, with consolidated calcareous sand underlying the mangrove peat and mud horizon. A stratigraphic core from a seaward mangrove edge (located between transects 4 and 5, Section 3.4, Fig. 3.1) showed 100 cm of mangrove mud above calcareous sand, and mud from 80-100 cm depth gave

a radiocarbon date of  $620 \pm 70$  years BP, a sediment accretion rate of about  $1.5 \text{ mm a}^{-1}$  over this period (Ellison, 2001). Undated cores offshore of the current mangrove seaward edge in the lagoon found mangrove mud underneath shallow calcareous lagoon sediment, indicating past dieback. The mangrove exists as a fringe along a sheltered estuarine embayment, is river-dominated in some areas, and tide-dominated in other sections. The entire mangrove is exposed to daily tides.

Nu'uuli receives drainage from a watershed contributing area (catchment area) of approximately 1,760 ha (Fig. 2.4). Six streams (Papa, Mata'alii, Sagamea, Sauino, Vaitele and an unnamed stream) drain into the mangrove. The sediment deposition in the more river-dominated parts of the system is likely positively correlated with the size of catchment area for each individual stream (Woodroffe, 1992). In the more tide-dominated areas of the wetland, which are distant from the six river mouths, tides cause sediment redistribution and resuspension. In these areas of the mangrove, sedimentation patterns are primarily controlled by marine processes. The tide-dominated areas are likely a net sink for organic matter due to the low-energy microtidal setting, and presence of dense roots, which dissipates tidal and wave energy. The eastern section of Nu'uuli mangrove is in a back-barrier geomorphic setting on the leeward side of a peninsula, receiving protection from wave energy. In one area of this eastern section of Nu'uuli mangrove, at the southern end of Coconut Point, is a low-productive dwarf mangrove setting, is tide-dominated, far from riverine influence, with sandy, nutrient-poor soil. In several areas upslope/landward of the mangrove landward margin there is a freshwater marsh, some of which is cultivated. Large portions of Nu'uuli mangrove have been filled for development since the early 1900's (Amerson et al., 1982; American Samoa Coastal Management Program, 1992; Scott, 1993). The mangrove exists in an enclosed estuarine embayment which has become increasingly constricted since construction of a runway in the early 1960s across a large proportion of the historic mouth of the bay, likely causing high water residence time in the bay, causing limited export from the bay. The average residence time for water in the bay is 30 hours, where 40% of the volume of the water in the bay is exchanged during a semidiurnal tidal cycle (Scott, 1993). Much of the landward mangrove



analysis of this core demonstrated that the swamp had been dominated by *Bruguiera* through this past period, as it is today.

Masefau mangrove receives drainage from a watershed contributing area of approximately 362 ha (Fig. 2.5), and there is relatively little development within the watershed. The entire mangrove is exposed to daily tides. The tide-dominated areas of the mangrove, located away from the stream channel and tidal creeks, are likely a net sink for organic matter due to the low-energy microtidal setting, and presence of dense roots, which dissipates tidal and wave energy. One stream (Talaloa stream) passes through the site, which transitions into an estuarine inlet from the ocean, with a complex network of tidal creeks distributed throughout the mangrove. Tree species Tahitian chestnut (*Inocarpus fagifera*) and beach hibiscus (*Hibiscus tiliaceus*) dominate along the relatively low salinity stream banks in the upper portion of the mangrove. The mangrove transitions into a freshwater marsh (an abandoned cultivated taro wetland) upslope/inland from the mangrove landward margin, extending inland until the terrain steepens and the stream becomes confined within narrow steep banks. A small mudflat, exposed at low tide, exists between the seaward mangrove edge and the mouth of the tidal creek where it enters the bay. Tidal exchange is through a narrow 15 m wide hardened inlet, underneath a concrete bridge over which the main coastal road of the village of Masefau passes. The majority of the area of the mangrove is tide-dominated, while areas proximate to tidal creeks and channels are river-dominated. Rapid deposition of terrigenous sediments may historically have resulted in microtopographic changes, where coastal progradation occurred with mangrove colonization of shoals at the river mouth, and replacement of mangroves by freshwater wetland systems landward (Woodroffe, 1987).



three springs discharge into the mangrove. The mangrove transitions into a freshwater marsh upslope/inland from the mangrove landward margin, of which large portions are cultivated for banana and coconut. A small mudflat, most of which is exposed at low tide, exists between the seaward mangrove edge and the mouth of the tidal creek where it then enters the bay. The mudflat largely lacks macrophytes (Scott, 1993). Tidal exchange is through a narrow 15 m - wide hardened inlet, underneath a concrete bridge over which the main coastal road of southern Tutuila passes. A large degree of development exists adjacent to and within the mangrove, including houses with seawalls, piggeries, chicken farms, and garbage dumps. Portions of the southern area of the mangrove have been filled for development, with seawall construction along the new mangrove landward margin. Inland from the mangrove is a small salt marsh with patches of marsh grass *Paspalum vaginatum* and candlebush *Cassia alata*, and a stand of *Hibiscus tiliaceus* trees (Scott, 1993). This site is in a similar environmental setting as Masefau but with a higher degree of development in its watershed.

In areas of the three study sites where river flow is the dominant flooding process, sediments are deposited from the river, while organic production is augmented by the freshwater input, and this organic matter is exported through river flow (Fleming et al., 1990; Furukawa et al., 1997; Woodroffe, 2002). At areas that are tide-dominated, there is bi-directional flux, transporting organic matter from the mangrove to adjacent coastal areas, but to a more limited extent than in the more riverine settings (Jiminez and Sauter, 1991; Lee, 1995, 1999; Woodroffe, 2002).

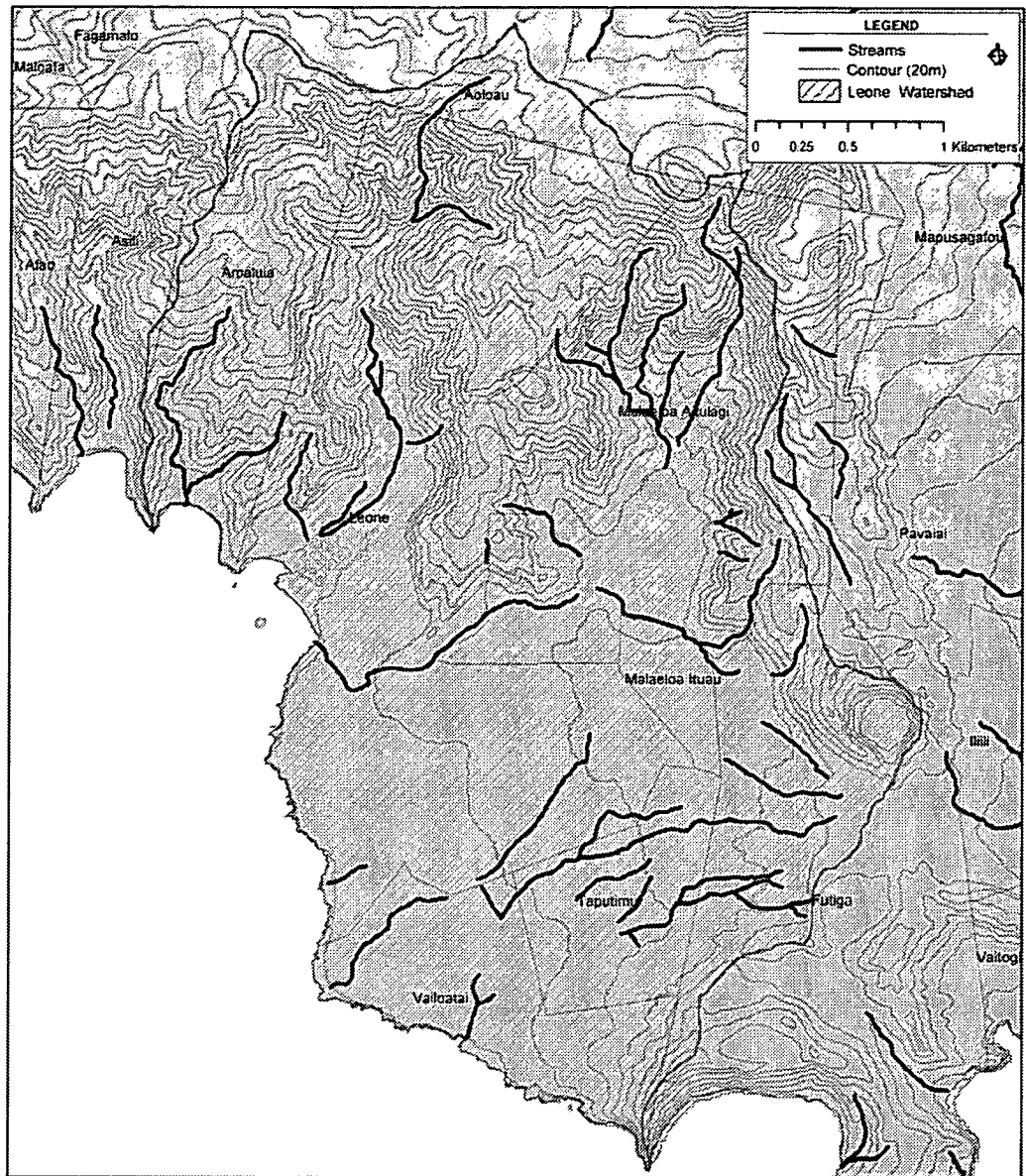


Fig. 2.6. Catchment, topography and streams of Leone mangrove, American Samoa.



## **Chapter 3**

### **Methods**

#### **American Samoa Mangrove Vulnerability and Responses to Relative Sea-Level Rise**

##### **3.1. PROJECTED TRENDS IN AMERICAN SAMOA RELATIVE SEA- LEVEL**

A linear temporal trend in regional relative sea-level for American Samoa was calculated using mean monthly relative sea-levels obtained from analysis of data from the Pago Pago, American Samoa tide gauge (located at  $-14^{\circ} 16' 59.9874''$ ,  $-170^{\circ} 40' 59.988''$ ). Data sources, corrected for changes in local datum, were the Permanent Service for Mean Sea Level and the University of Hawaii Sea Level Center Joint Archive for Sea-Level and GLOSS/CLIVAR Research Quality Data Set databases. A linear regression model was fit to the mean monthly relative sea-level data from October 1948 through May 2004, an elapsed period of 55.6 years.

The observed American Samoa mean regional relative sea-level rise trend was compared to the globally calculated rate of past sea-level change determined by the Intergovernmental Panel on Climate Change (IPCC) (Church et al., 2001). The global sea-level rise minimum and maximum projections through the year 2100 were then applied to the American Samoa observed rate of change in relative sea-level to determine a range of relative sea-level projections for American Samoa through the year 2100. IPCC models A2 and B1 provide the minimum projections for the change in global mean sea level, and models A1T and A1FI provide the maximum projections (Appendix II.5, Table II.5.1 in Church et al., 2001).

The IPCC range of projections for change in global sea-level were used for American Samoa's projections for regional relative sea-level rise because observed American Samoa rate of regional relative sea-level change during the 20<sup>th</sup> century (the same period of observation used by IPCC) (October 1948 through December 2000) of  $1.77 \text{ mm a}^{-1}$  ( $1.41 - 2.12$

95% CI, N = 581) over the observed 51.83 years (based on fitting a linear regression model to mean monthly relative sea-levels) is within the IPCC's (Church et al., 2001) and Church et al.'s (2004a) uncertainties for the rate of change of global average sea-level. The IPCC's best estimate of global average sea level change during the 20th century, based mainly on tide gauge observations, is  $1.5 \pm 0.5 \text{ mm a}^{-1}$  (Church et al., 2001; Cazenave and Nerem, 2004), while Church et al. (2004a) provide an estimate of  $1.8 \pm 0.3 \text{ mm a}^{-1}$  from 1950 – 2000.

At the time of conducting this research, results from the IPCC Third Assessment Report (Church et al., 2001) were available and used. The IPCC Fourth Assessment Report was subsequently released (Solomon et al., 2007). For each scenario for projected global sea level rise in the Fourth Assessment Report, the midpoint of the range is within 10% of the Third Assessment Report, however, the ranges in the Fourth Assessment Report are narrower than in the previous assessment mainly due to improved information about some uncertainties in contributions to each projection (Solomon et al., 2007).

### **3.2. TRENDS IN FREQUENCY AND ELEVATION OF EXTREME HIGH WATER EVENTS**

Analysis was conducted to determine if there have been trends in change in the frequency and elevation of extreme high water events in American Samoa over the past 55 years, and to determine if increased extreme water levels exceed that which can be explained by factors causing observed rise in mean and median relative sea-level, such as due to increased intensity of storms. Extreme high water event levels are projected to increase over coming decades as a result of the same atmospheric and oceanic factors that are causing global sea-level to rise, and possibly also as a result of other influences, such as variations in regional climate, through change in storminess (Hunter, 2002; Woodworth and Blackman, 2004; Church et al., 2001, 2004b).

Data sources from the Pago Pago, American Samoa tide gauge, corrected for changes in local datum, were again the Permanent Service for

Mean Sea Level and University of Hawaii Sea Level Center Joint Archive for Sea-Level and GLOSS/CLIVAR Research Quality Data Set databases. It appears that, prior to 1975, tide gauge sea level data were recorded in Imperial units and later converted to metric units, giving a data resolution of about 30 mm, where intervals of 0.1 foot were converted to 30 mm intervals. Resolution improved as tide gauge equipment modernized. At the time of conducting this analysis, hourly Pago Pago tide gauge data were available from 11 am, 8 September 1948 through 11 pm, 30 September 2004, however, hourly tide gauge records for 1948 and 2004 are less than 75% complete and were therefore not included in analyses for this study component.

Percentile time series analysis (Woodworth and Blackman, 2004; Hunter, 2002) was employed to study changes in extreme high water levels above changes in relative mean sea-level. For a year with no data gaps, there are 8760 hourly sea-level measurements in a non-leap year and 8784 in a leap year. The data were sorted in ascending order to compute percentile levels for the year. Percentile values were calculated at 50, 99.9, and 99.95 percentiles. The 50-percentile is the median, which corresponds well to mean relative sea-level at sites with relatively weak tidal regimes (Hunter, 2002; Woodworth and Blackman, 2004). The sea levels that are  $\geq$  the 99.9 percentile correspond approximately to the levels of the 8 highest hourly sea-level values, and sea levels that are  $\geq$  the 99.95 percentile correspond approximately to the 4 highest hourly sea-level values in both leap and non-leap years. The individual percentiles were reduced to median relative sea-level to remove the common signal by subtracting that year's 50-percentile.

The change in frequency of extreme high water events was calculated first by determining the 99.9 and 99.95 percentiles of median annual relative sea-level values from 1949-2003. The 99.9 and 99.95 percentiles were calculated by (i) sorting the hourly tide gauge relative sea-level data for each individual year from lowest to highest observed sea-level; (ii) determining the 50 percentile value for that year; (iii) reducing the observed hourly sea-levels to median relative sea-level by subtracting the year's 50-percentile, so that the resulting data are the elevation in mm above that year's median sea-

level; (iv) determining the 99.9 and 99.95 percentiles for each year reduced to that year's median sea-level; and (v) determining the 99.9 percentile reduced to median sea-level by combining all annual 99.9 percentile values reduced to median sea-level and sorting them into ascending order to identify the median value, and conducting the same process to identify the 99.95 percentile reduced to median sea-level.

A single extreme high water "event" at or above the 99.9 percentile was defined for this analysis as using all observed hourly sea levels occurring within a single one-week period which are  $\geq$  the 99.9 percentile value. The hour of the onset of an extreme high water event was used as the date of that event. A similar definition was used for an extreme high water event at the 99.95 percentile level.

The trend in frequency of extreme high water events was presented in four ways. (i) A linear regression model was fit to a plot of year versus number of extreme high water events to determine a trend in frequency of extreme high water events over a 55-year period from 1949-2003. (ii) For each year, the elapsed return times between the initiations of extreme high water events, or "return periods", were calculated. A linear regression model was fit to a plot of year versus the return time between extreme high water events to see if the recurrence of these high water events is changing over time. (iii) A comparison of the number of extreme high water events occurring in the first half of the dataset versus the second half was also conducted. And (d) a comparison of the mean time between extreme high water events occurring in the first half of the dataset versus the second half was also conducted. The midpoint of the data series, the middle of 1976, happens to coincide with the approximate timing of a possible sudden climate shift (Graham, 1995).

The trend in absolute elevations of extreme high water events relative to the tide gauge datum was determined by (i) sorting the hourly tide gauge sea-levels for each year from lowest to highest, (ii) determining the 99.95 percentile value for that year (not reduced to each year's median), (iii) plotting each year's sea-level values  $\geq$  the 99.95 value for that year versus time, and (iv) fitting a simple linear regression model to the data. A similar plot of each year's sea-levels  $\geq$  the 99.9 percentile versus time was also

made. Plots of mean and median sea levels versus time were also produced and linear regression models fit to the data series to show the trend in change in mean and median sea-levels to compare to the trend in change in extreme high water event levels.

A second method was employed to observe if the same or different factors that cause change in mean and median sea-levels are causing changes in extreme high water levels to the same degree. (i) The hourly data for each year were sorted and reduced to median relative sea-level by subtracting the year's 50-percentile so that the data now represent elevations relative to that year's median sea-level, (ii) the new 99.95 percentile now reduced to median sea-level is determined for each year, (iii) the sea-levels for each year  $\geq$  each individual year's 99.95 percentile reduced to median sea-level were plotted against time and a linear regression model was fit to the data to determine a slope. A similar plot of each year's sea-levels  $\geq$  the 99.9 percentile reduced to median sea-level versus time was also made.

### **3.3. HISTORICAL RECONSTRUCTION OF MANGROVE MARGIN POSITION**

Positions of seaward margins of Nu'uuli, Masefau and Leone mangroves were reconstructed by analyzing a time series of remotely sensed imagery and GIS techniques. Aerial photos showing the seaward boundary of Masefau mangrove are available from 1961, 1971, 1990, and 1994, for Nu'uuli mangrove are available from 1961, 1971, 1984, 1990, and 1994, and for Leone mangrove are available from 1961, 1966, 1971, 1984, 1990, and 1994. Ikonos satellite images from 2001 and QuickBird satellite images from 2003-2004 are also available for the three study sites. The IKONOS and QuickBird satellite imagery were geo-referenced to the UTM NAD83 Zone 2 South HARN projection and coordinate system. ERDAS Imagine 8.7 software was used to co-register the aerial photos to the georeferenced 2001 Ikonos satellite imagery. A minimum of twenty ground control points were used per aerial photo for co-registration. A third order polynomial model was used to co-register the aerial photos. Appendix 2 provides metadata for the

historical aerial photos and satellite images of the mangrove study sites used in these analyses.

The mangrove seaward margins and margins of major tidal creeks for the three study sites were identified and digitized from each co-registered aerial photo and satellite image. The seaward margin was defined as the unbroken canopy edge, thus excluding opportunistic, pioneer mangrove vegetation. ArcGIS software was used to calculate the area between the seaward mangrove margin and a fixed line seaward of the mangrove (information used in Section 3.9.1 and Equation 4) for each remotely sensed image. Change in the area of open water between the seaward mangrove margin and the fixed line is caused by change in position of the mangrove seaward margin. It was not possible to identify the position of the landward mangrove margins from interpretation of the remotely sensed images because it was not possible to accurately differentiate between the signatures of mangrove versus adjacent upland forest canopy cover. The length of each study site's seaward margin was measured for each historical image using Arc GIS. Mean length of seaward margins and observed changes in mangrove area from movement of the seaward margin were used to estimate the distance the margin moved over the observed period.

### **3.4. TRENDS IN MANGROVE SURFACE ELEVATION**

#### **3.4.1. Sampling Design**

Spatial and temporal trends in sediment surface elevation, accounting for the full sediment profile (Fig. 1.8), were estimated for Nu'uuli and Masefau mangroves. Change in mangrove sediment surface elevation was measured using a hierarchical sampling approach comprising ten poly vinyl chloride (PVC) stakes deployed within each of 20 sampling stations at Nu'uuli (Table 3.1, Fig. 3.1) and ten stakes deployed within each of 13 sampling stations at Masefau (Table 3.2, Fig. 3.2).

Table 3.1. Coordinates for the center of elevation stake stations, and distances and compass headings along transects from the landward mangrove margin, Nu'uuli mangrove.

Transect	Stake Station	Latitude (S)	Longitude (W)	Heading from Landward Margin	Distance from Landward Margin (m)
1	a	14° 19' 0.6"	170° 42' 48.6"	142°	14
1	b	14° 19' 1.9"	170° 42' 48.8"	142°	54
2	a	14° 18' 49"	170° 42' 45"	155°	43
2	b	14° 18' 50"	170° 42' 45"	155°	73
2	c	14° 18' 53"	170° 42' 44"	155°	169
2	d	14° 18' 55"	170° 42' 44"	155°	230
2	e	14° 18' 56"	170° 42' 43"	155°	266
3	a	14° 18' 37.5"	170° 42' 26.6"	154°	20
3	b	14° 18' 38.9"	170° 42' 25.8"	154°	70
3	c	14° 18' 40.6"	170° 42' 25.7"	154°	120
4	a	14° 18' 40.6"	170° 42' 11.6"	216°	20
4	b	14° 18' 42.3"	170° 42' 12.2"	216°	75
4	c	14° 18' 43.2"	170° 42' 13.4"	216°	117
5	a	14° 18' 50.5"	170° 42' 6.2"	282°	16.5
5	b	14° 18' 50.9"	170° 42' 7.3"	282°	52
6	a	14° 18' 59.4"	170° 42' 5.9"	270°	8
6	b	14° 18' 59.0"	170° 42' 7.9"	270°	69
7	a	14° 19' 13.3"	170° 42' 9.5"	284°	6
8	a	14° 19' 14.8"	170° 42' 11.6"	349°	25
8	b	14° 19' 14.3"	170° 42' 11.7"	349°	41

1

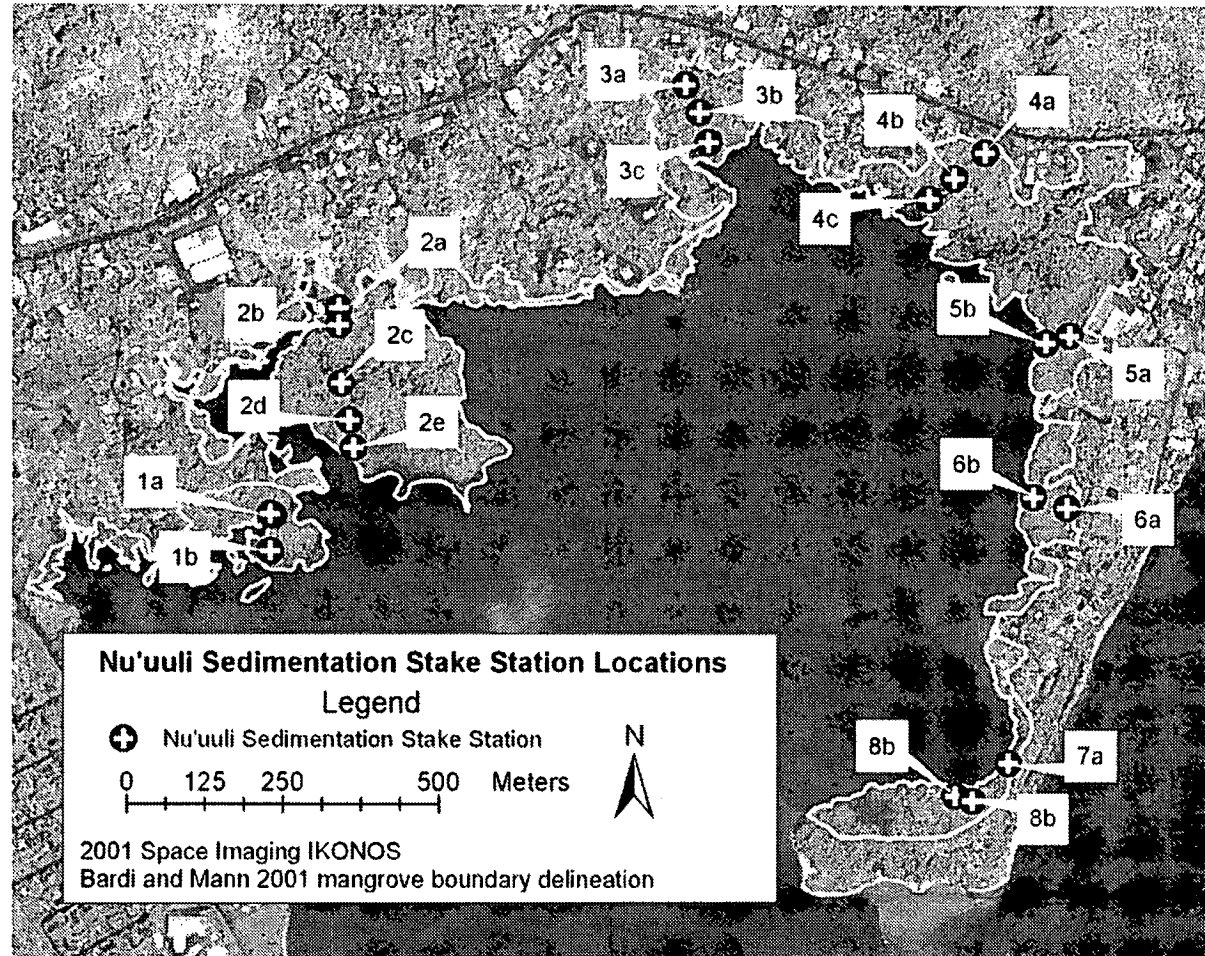


Fig. 3.1. Location of elevation stake stations along transects in Nu'uuli mangrove.



Table 3.2. Coordinates for the center of elevation stake stations, and distances and compass headings along transects from the landward mangrove margin, Masefau mangrove. The seaward margin for both transects is located at 170° 38' W, 14° 15' S.

Transect	Stake Station	Latitude (S)	Longitude (W)	Heading from Seaward Margin	Distance from Seaward Margin (m)
1	a	14° 15' 19.3"	170° 37' 56.2"	218°	11
1	b	14° 15' 20.0"	170° 37' 57.8"	218°	62
1	c	14° 15' 20.7"	170° 37' 58.3"	218°	87
1	d	14° 15' 22.3"	170° 37' 59.5"	218°	146
1	e	14° 15' 22.3"	170° 38' 01.0"	218°	181
1	f	14° 15' 24.2"	170° 38' 02.2"	218°	245
1	g	14° 15' 25.0"	170° 38' 03.2"	218°	284
2	a	14° 15' 21.9"	170° 37' 57.7"	199°	103
2	b	14° 15' 23.4"	170° 37' 58.8"	199°	159
2	c	14° 15' 24.5"	170° 38' 0.4"	199°	214
2	d	14° 15' 26.3"	170° 38' 0.8"	199°	267
2	e	14° 15' 27.7"	170° 38' 01.7"	199°	317
2	f	14° 15' 27.9"	170° 38' 02.2"	199°	331

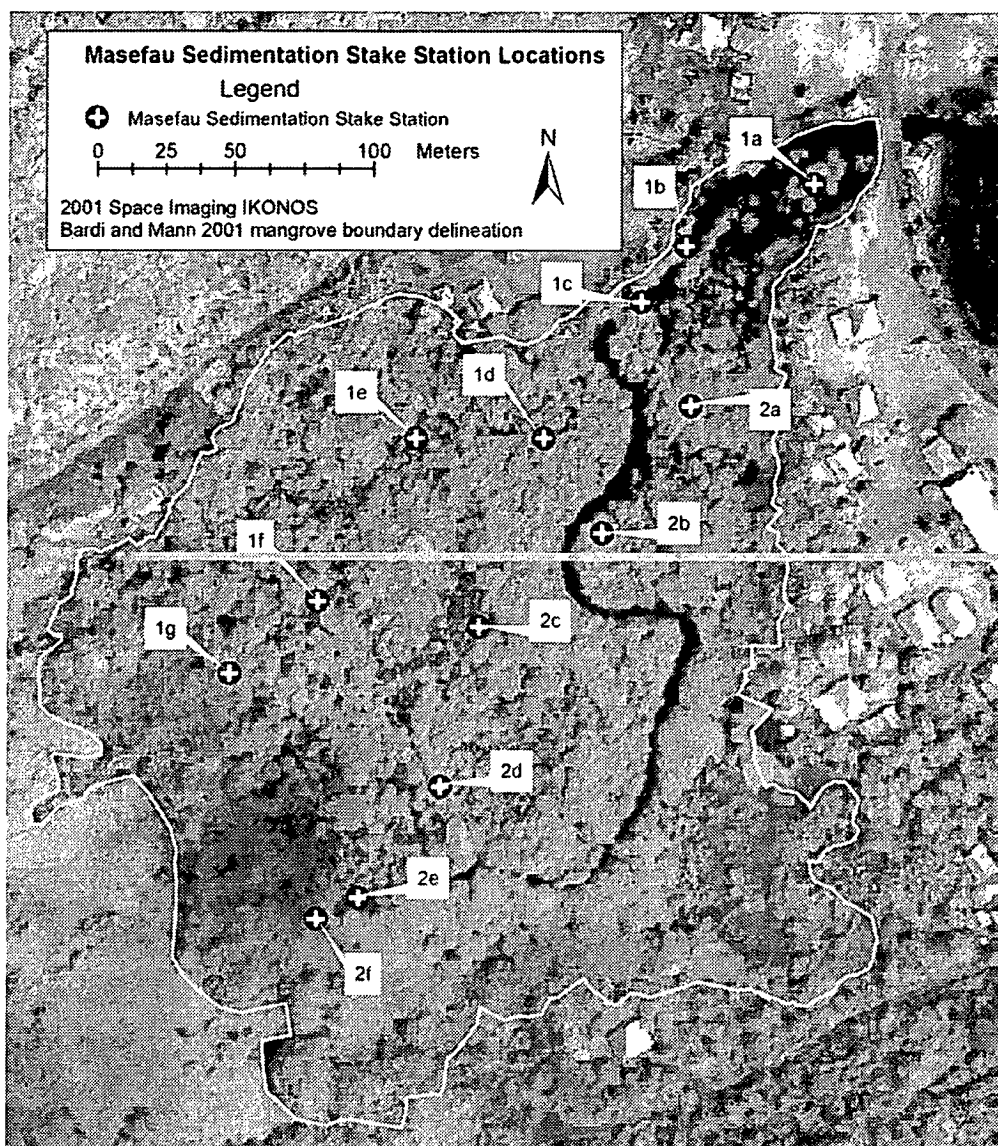


Fig. 3.2. Location of elevation stake stations along transects in Masefau mangrove.

The 330 stakes were located along ten perpendicular seaward-landward transects located across the major mangrove sections (Figs. 3.1 and 3.2). Transects in both study sites were positioned to cut across vegetation zones by being situated along the gradient from the high water line to lower elevations. Nu'uuli transects were located in the center of major mangrove subsections to reduce edge effects and to ensure that the sites were sufficiently sampled so the resulting data characterize the entire system. Transects in Masefau were located starting at the single seaward

margin point, with one transect located through the center of the wetland generally along the lowest gradient, and one additional transect fanning out to the East and adjacent to the central transect. Stakes were located in stations of 10 replicates every 50 m along transects, with replicates of 5 stakes located on either side of the transect every 2 m perpendicular to the transect, with the central two stakes of the station each 2 m from the transect (4 m apart) to minimize disturbance from researchers walking along the transect. During the study period, 34 of the stakes were lost (29 from Nu'uuli and 5 from Masefau), possibly a result of human disturbance.

PVC stakes were placed into the sediment leaving 15 cm of the stake protruding from the sediment surface, enough so that the stake is unlikely to become buried over coming years, but not so much as to make the stake overly visible to people who might disturb the stakes. Stakes were 1.5 or 2.0 cm in diameter, 1.5 m long, but were cut shorter in areas where the depth of consolidated substrate was shallower than 1.35 m and prevented inserting the full length of the stake. Eighty five percent (279) of the stakes were inserted until they reached consolidated substrate. The remaining 51 stakes did not penetrate the mangrove peat layer to reach hard substrate, but were inserted 1.35 m into the substrate, below mangrove live roots (e.g., Gill and Tomlinson, 1977; Tomlinson, 1986) so that the stakes' vertical position was not affected by factors controlling sediment elevation occurring within the upper 1.35 m of sediment. For stakes inserted through the entire soil profile, this method captured net change in elevation of the mangrove sediment surface, including from sediment accretion and erosion, leaf litter accumulation, benthic mat production, root production, autocompaction, fluctuations in water table levels and pore water storage (Section 1.5.4, Table 1.3) (Lynch et al., 1989; Donnelly and Bertness, 2001; Woodroffe, 1992, 2002; Krauss et al., 2003; Rogers et al., 2005; Whelan et al., 2005; Cahoon et al., 1995, 1999, 2003, 2006, McKee et al., 2007).

The locations of the center of stake stations were determined using Leica SR399 dual frequency and Garmin 72 GPS units, with horizontal accuracies on the order of centimeters and tens of meters, respectively. The vertical position of individual stakes from the GPS units was not used

because canopy cover and intermittent satellite coverage prevented achieving vertical accuracies of better than  $\pm 2$  cm.

The change in sediment surface elevation at each stake was then measured as  $\pm 1$  mm from the top of the stake to the wetland surface at five sampling occasions in Nu'uuli, and four occasions in Masefau.

Measurements were made over 1.53 years in Nu'uuli mangrove, from 25 August 2004 to 6 March 2006, and over 1.30 years in Masefau mangrove, from 17 November 2004 to 8 March 2006. The heights of the top of the stakes above the sediment surface were recorded using a standard folding ruler. The measurements of elevation from the top of the stakes to the wetland surface were taken from 10 cm away from the stake, using a level to keep horizontal with the top of the stake, to avoid any distortion to the wetland surface caused by the presence of the stake, such as from scouring around the base of the stake. In Nu'uuli, the elevations were measured  $90^\circ$  to the right of the stake when the researcher is facing the mangrove seaward margin along a transect that connects stations, and in Masefau, when the researcher is facing the landward margin. This was done to minimize variance between measurements at the same stakes at different times resulting from variation in microtopography of the mangrove surface and to maximize consistency in measurements made by three researchers. Measurements were attempted to be made only at low tide to facilitate locating the stakes and to reduce error from identifying the mangrove surface when tidal water suspends the fine sediment from the mangrove surface.

The sampling design therefore comprises a hierarchy of three sampling levels: (i) stakes were nested within (ii) stations, which were nested within (iii) study sites. Stakes were surveyed between August 2004 and March 2006. The sampling stakes and stations at each site are random effects, while the two study sites are considered fixed effects. Stake and station locations are considered a random sample from the large number of possible locations. Overall, the study comprised 902 samples (total number of stake elevation measurements) for the Nu'uuli study site and 510 for the Masefau site (overall  $N = 1,412$ ).

### 3.4.2. Statistical approach

Spatial and temporal trends in mangrove sediment elevation for the 1,412 samples were estimated using a nonparametric regression modeling approach, known as generalized additive modeling, which allows (i) flexible specification of the error and link functions, and (ii) arbitrary specification of the functional form for each predictor included in the model (Hastie and Tibshirani, 1990). Generalized additive regression models (GAMs) are the preferred choice for analysis of spatial and temporal trends in data series that may comprise nonlinear behavior and involve multi-level sampling designs (Fahrmeir and Lang, 2001; Wood, 2006). The GAM approach relaxes normality assumptions of standard linear regression modeling approach and supports flexible link specification while the functional form (linear, nonlinear) for each predictor is estimated from the data using nonparametric smoothers while conditioning on all other covariates included in the model.

A GAM therefore remains additive (unlike nonlinear regression) and hence simple to interpret by comprising a separate function for each predictor in the regression model, similar to a standard multiple linear regression model (Hastie and Tibshirani, 1990). Interaction terms can be implemented in a GAM using an approach where a covariate is nested within each level of a factor and is known as a varying coefficient GAM or VCGAM (Hastie and Tibshirani, 1993). GAMS and VCGAMS are readily extended to account for random effects (Wood, 2006) that arise in studies that involve multiple measurements on the same sampling unit, such as in this study, which involved repeated measurements for 10 PVC elevation stakes within the sampling stations at each of the two study sites. Random effects GAMs are known as generalized additive mixed models (GAMMs) as they comprise both fixed effects and random effects (Fahrmeir and Lang, 2001; Wood, 2006).

Specifically, a random effects VCGAM (or a VCGAMM) was adopted here to estimate spatial and temporal trends in the sediment surface elevation data. The GAMM approach enables the heterogeneity of each stake and sampling station within the two sites to be taken into account in the temporal trend analysis. Separate time-specific trends were fitted for each level of the site factor (Nu'uuli, Masefau) and so is a VCGAMM. This

nonparametric regression model was fitted using thin plate regression splines to model any nonlinear temporal effects, identity link, quasi-likelihood error structure, and with all the smoothness parameters determined using generalized cross-validation (Wood, 2006). The spline smoothers in the nonparametric regression track any linear or nonlinear trend in the data without having to assume any specific functional form, unlike a conventional linear or nonlinear regression model (Hastie and Tibshirani, 1990). A quasi-likelihood error function included in the nonparametric regression model is quite general and also precludes a need to assume any specific parametric error distribution (unlike a standard linear regression model) and depends only on a mean-variance relationship derived from the data (McCullagh and Nelder, 1989). All models were fitted using the *mgcv* package (Wood, 2006) available for the open source statistical modeling program R (Ihaka and Gentleman, 1996).

Ten measurements of the elevation of a single stake were repeated at a single time by one researcher to provide an indication of the precision (uniformity) of the measurement method. A second researcher made ten measurements of the elevation of the same stake at the same time to provide an indication of the replicability of the measurement method. The potential measurement error attributable to the different researchers is explicitly accounted for through the use of the random effects regression modeling approach employed here.

### **3.5. MANGROVE SEDIMENT ACCRETION RATES FROM $^{137}\text{Cs}$ AND EXCESS $^{210}\text{Pb}$ ACTIVITY DATING**

Accretion rates in in Nu'uuli and Masefau mangroves were measured through gamma ray spectrometry of sediment profiles for depth of sediment above the  $^{137}\text{Cs}$  peak horizon and geochronology from activity of excess  $^{210}\text{Pb}$ . Core extraction locations are shown in Figs. 3.3 and 3.4. The core from Nu'uuli was taken at 170° 42' 44.3" W, 14° 18' 51.8" S, and the core from Masefau was taken at 170° 38' 2.9" W, 14° 15' 25.3" S within  $\pm 15$  m. The locations selected to take the cores were selected to avoid anomalies such as proximity to a source of sediment redistribution, erosion, accretion, or

compaction, such as next to a tidal creek, stream, river, trail, piggeries, agricultural plots, or other human activity, or the edge of the mangrove wetland (e.g., Lynch et al., 1989). This method provides information on the influence of processes within the upper soil profile on the mangrove sediment surface elevation (Fig. 1.8).

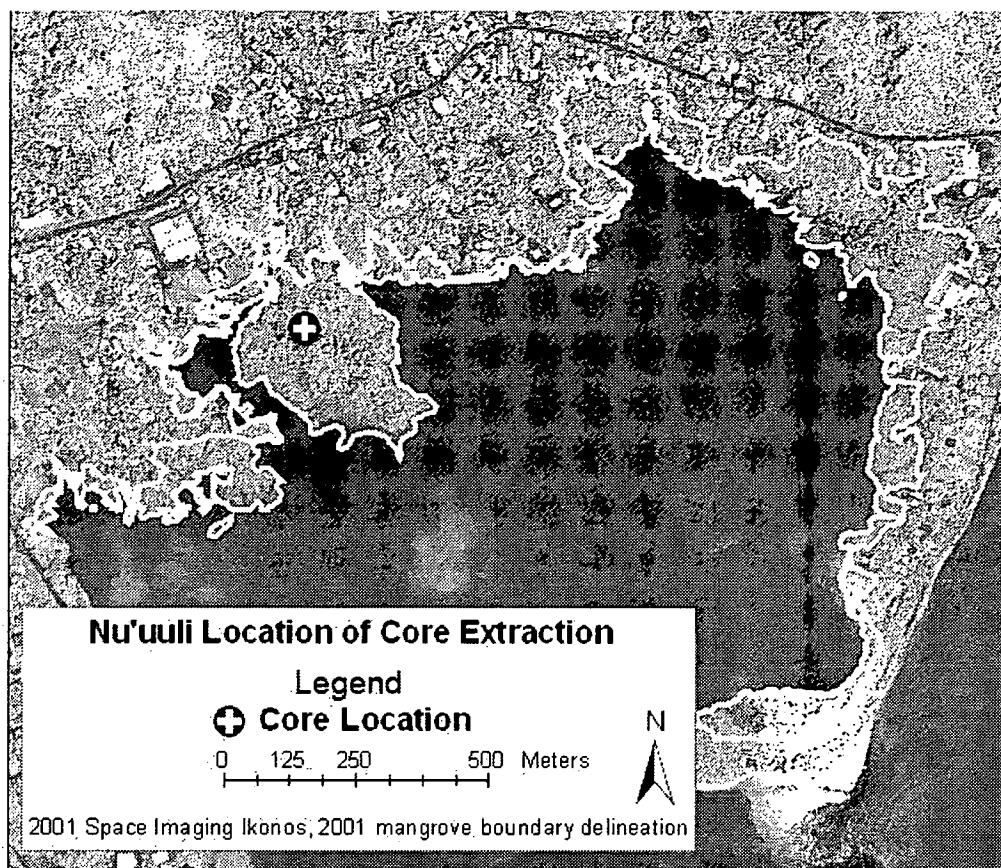


Fig. 3.3. Core extraction location, Nu'uuli mangrove, for  $^{137}\text{Cs}$  and excess  $^{210}\text{Pb}$  analysis to measure sediment accretion rates.



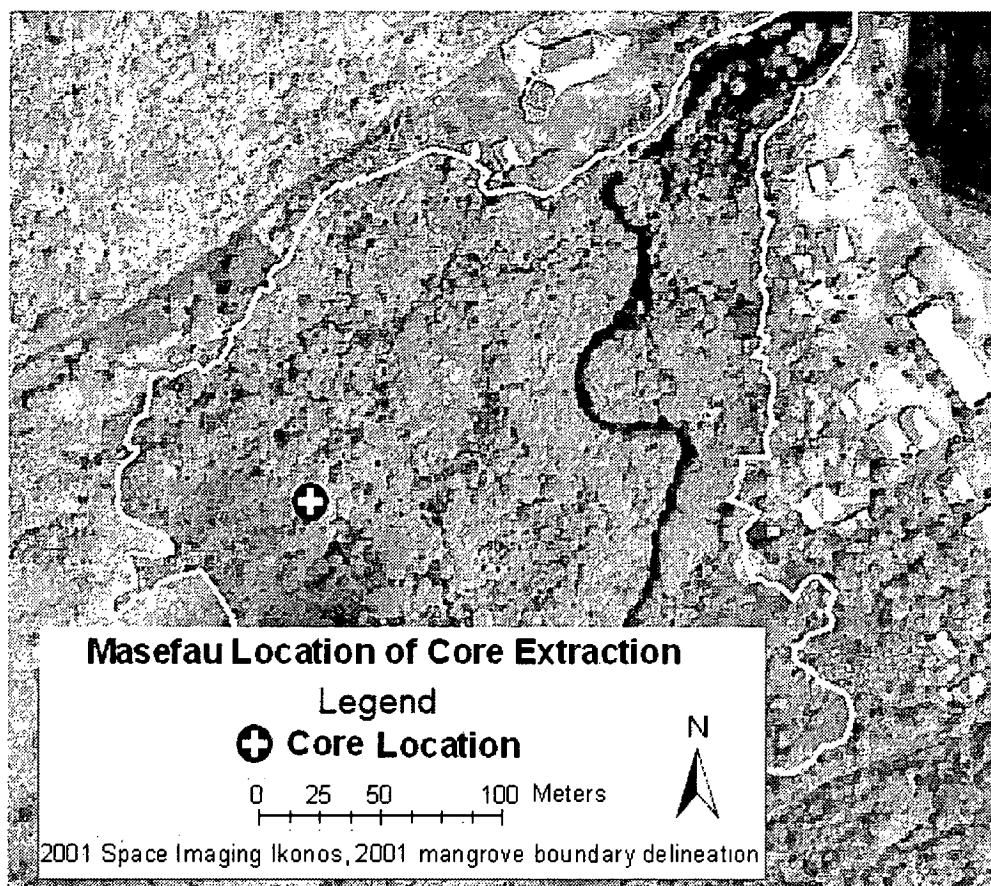


Fig. 3.4. Core extraction location, Masefau mangrove, for  $^{137}\text{Cs}$  and excess  $^{210}\text{Pb}$  analysis to measure sediment accretion rates.

### 3.5.1. Radioisotopic Analyses

Radioisotopic studies of sediments from each study site were undertaken to investigate the sediment accumulation history. The University of Hawaii Radioanalytical Facility oven-dried the samples at  $60^{\circ}\text{C}$  for 48 hours, ground them with mortar and pestle, mixed them, and sealed the samples in plastic counting vials with high-vacuum resin. The samples were then incubated for 21 days to allow ingrowth of  $^{222}\text{Rn}$  and daughter activities before counting. High-purity Ge gamma spectrometry techniques were used to measure the radioisotope activities in the stratigraphic subsamples from the two cores. A Ge detector (EG&G Ortec Gamma-X), extended-range, high-purity Ge detector with 4096-channel analyzer and computer control was used to conduct gamma-ray spectrometry to detect  $^{137}\text{Cs}$  and excess  $^{210}\text{Pb}$  specific activities, after methods described by McMurtry et al (1995). Sediment

accretion rates are determined from a semi-log excess  $^{210}\text{Pb}$  decay plot based on a half-life distance calculation from the least-squares fit to the data, and from the depth of the  $^{137}\text{Cs}$  peak horizon.

#### **3.5.1.1. Depth of sediment above $^{137}\text{Cs}$ peak horizon and horizon of first occurrence**

Cesium-137, an artificial radionuclide produced by nuclear fission with a half-life of 30.17 years, is distributed across the earth's surface due to fallout from atmospheric nuclear tests and accidental releases from nuclear reactors (International Atomic Energy Agency, 1998). Widespread global distribution of  $^{137}\text{Cs}$  began with the first atmospheric atomic explosion in Hiroshima in 1945, with subsequent atmospheric testing of high-yield fission nuclear weapons during the 1950s and 1960s, and the last in October 1980 in China (Agudo, 1998). 1963 is the approximate year of maximum cesium fallout in the southern hemisphere (Agudo, 1998) and 1954 is the year of first significant  $^{137}\text{Cs}$  fallout (DeLaune et al., 1978; Lynch et al., 1989; Agudo, 1998; Goodbred and Kuehl, 1998). The thickness of sediment above the sediment horizon with the  $^{137}\text{Cs}$  peak can be used to estimate the average sediment accretion rate since the deposition of this horizon in 1963. This method has been used in several erosion and accretion studies in wetlands, other aquatic habitats, and terrestrial environments (e.g., DeLaune et al., 1978; Lynch et al., 1989; Agudo, 1998; Goodbred and Kuehl, 1998; Donnelly and Bertness, 2001).

This technique assumes that the depth of  $^{137}\text{Cs}$  particles contributing to the 1963 peak and 1954 year of first fallout are solely due to subsequent accretion above this and not other processes. This method further assumes that there was a relatively short residence time of  $^{137}\text{Cs}$  in the water column of less than one year before deposition to the mangrove sediment surface (McMurtry et al., 1995). However, mixing of the sediment by biotic and abiotic processes can alter the depth and resolution of the horizon of peak  $^{137}\text{Cs}$  concentration and first detection. Furthermore, direct fallout may not be the sole source of  $^{137}\text{Cs}$ , as there may be intermediate reservoirs between the atmosphere and the mangrove sediment, such as watershed soils and lakes, which can lead to inaccurate analysis of the  $^{137}\text{Cs}$  depositional record

if there has been a gradual release of  $^{137}\text{Cs}$  to the mangrove sediment (McMurtry et al., 1995). Even at sites where intermediate reservoirs are primary sources of  $^{137}\text{Cs}$ , an accurate sedimentation rate can be estimated from the level of first  $^{137}\text{Cs}$  detection.

Desorption and diffusion of  $^{137}\text{Cs}$  could affect the sedimentation rate estimate, as this can cause  $^{137}\text{Cs}$  to penetrate deeper in the sediment horizon than the actual solid accumulation thickness, however this is not believed to be a cause of major error based on the results of a simple diffusion model (McMurtry et al., 1995). Cesium mobility is not typically problematic for measuring  $^{137}\text{Cs}$  activity in mangrove sediment, as this particle-reactive cation is strongly adsorbed onto the cation exchange sites of expandable clays, where it becomes mostly fixed (Delaune et al., 1978; McMurtry et al., 1995; Goodbred and Kuehl, 1998). However  $^{137}\text{Cs}$  has low uptake onto organic matter (McMurtry et al., 1995), and substantial  $^{137}\text{Cs}$  diffusion has been observed in marine calcareous muds with little detrital clay in the Marshall Islands.

#### **3.5.1.2. Activity of excess $^{210}\text{Pb}$**

Lead-210 radioisotope geochronology has been used widely in wetlands, rivers, estuaries, and lakes to determine recent (i.e. 10-150 years) rates of sediment accretion (e.g., Lynch et al., 1989; McMurtry et al., 1995; Goodbred and Kuehl, 1998; Donnelly and Bertness, 2001). The assumption is that radioisotope activity in excess of the level of atmospheric fallout is derived from sediment accretion.

$^{210}\text{Pb}$  activity in sediments is derived from the decay of its parent  $^{226}\text{Ra}$ , referred to as “supported”  $^{210}\text{Pb}$  activity (Lynch et al., 1989).  $^{226}\text{Ra}$  within the crust decays to  $^{222}\text{Rn}$  (radon gas). A fraction of the  $^{222}\text{Rn}$  enters the atmosphere.  $^{222}\text{Rn}$  has a short, 3.2 day, half-life, and decays to  $^{210}\text{Pb}$  once in the atmosphere, which is then deposited to the earth’s surface in precipitation (Lynch et al., 1998; Donnelly and Bertness, 2001). The  $^{210}\text{Pb}$  input to the sediment from atmospheric deposition is referred to as “unsupported” or “excess” lead activity (Lynch et al., 1989). The half-life of  $^{210}\text{Pb}$  is 22.3 years, and the decay of the excess  $^{210}\text{Pb}$  is used to determine sediment accretion rates (Lynch et al., 1989). This method assumes there is

uniform and constant input of lead to the sediment surface (there is a constant accretion rate), and continual burial of the nuclide (Lynch et al., 1989; Goodbred and Kuehl, 1998). If these assumptions are met, then there will be an exponential decrease in  $^{210}\text{Pb}$  activity with depth that can be used to estimate sediment accretion rates (Donnelly and Bertness, 2001).

The  $^{210}\text{Pb}$  activity at the depth where the sediment profile becomes asymptotic is assumed to be the supported  $^{210}\text{Pb}$  level (the amount of  $^{210}\text{Pb}$  produced from the decay of  $^{222}\text{Rn}$  within the sediment column and not deposited from the atmosphere). An average supported  $^{210}\text{Pb}$  value is calculated and subtracted from  $^{210}\text{Pb}$  activity values in the profile above where the curve becomes asymptotic, providing a curve of unsupported lead activity in the sediment profile. An accretion rate is calculated using the equation:

$$A_x = A_0 e^{-(\lambda x/S)}$$

where  $A_x$  is the unsupported  $^{210}\text{Pb}$  activity at depth  $x$ ,  $A_0$  is the unsupported  $^{210}\text{Pb}$  activity in the modern surface layer of the sediment profile,  $\lambda$  is the  $^{210}\text{Pb}$  decay constant of  $-0.03114 \text{ a}^{-1}$ ,  $x$  is depth, and  $S$  is the sedimentation rate (Donnelly and Bertness, 2001).

### 3.5.2. Coring

Compression during extraction of sediment cores can be minimized by using a wide diameter, thin-walled tube corer. Goodbred and Kuehl (1998) used a 15 cm diameter PVC push core to collect cores from saturated sediment of a river floodplain and delta complex for analysis of  $^{137}\text{Cs}$  activity. Delaune et al. (1978) and Lynch et al. (1989) used a 15 cm diameter, thin-walled, 0.5 m long aluminum tube corer to collect sediment cores from a salt marsh, lake, and mangrove wetlands to measure  $^{137}\text{Cs}$  activity. Previous studies have used 2 cm core sections or larger to observe  $^{137}\text{Cs}$  activity (Delaune et al., 1978; Lynch et al., 1989; Goodbred and Kuehl, 1998), however smaller, e.g., 1 cm sections, can be used to produce higher spatial resolution.

Two cores were taken on 9 June 2004, one from each of the two study sites (Nu'uuli and Masefau) to determine the depth of sediment above the

peak  $^{137}\text{Cs}$  horizon and measure excess  $^{210}\text{Pb}$  activity, providing estimates of the recent sediment accretion rate. A 75 cm long, 15 cm internal diameter PVC pipe was used as a corer. The outside wall of the bottom of the pipe was planed to make a cutting blade. Compression of the core profile was measured after the corer was inserted into the sediment but before extracting the core, by comparing the height of the top of the corer above the sediment surface measured on the outside of the corer to the depth inside the corer from the top of the corer to the sediment surface.

The corer was dug out of the ground using a shovel, and a plate was placed over the bottom end of the corer to prevent the profile from falling out of the corer as it was lifted from the hole. The profile was gradually extruded from the corer bottom first onto a clean sheet of plastic. The core was sectioned into 1 cm increments and placed into soil samples bags in the field. The core was cut using a 20 cm long, serrated bread knife, washed clean after cutting each section to minimize contamination between sections. A total of 38 1-cm-thick sections were recovered from each of the two cores.

### **3.6. RELATIVE SEA-LEVEL CONTROL ON MANGROVE POSITION**

Linear regression analysis was conducted to determine if there is a significant positive correlation between the change in mangrove area caused by movement of the seaward mangrove margin and change in relative sea-level. This was conducted to assess if change in site-specific and regional relative sea-level has been a primary control on observed changes in mangrove position. This assessment did not account for changes in mangrove area that may have occurred from movement of the landward mangrove margin.

Three methods were used for this assessment, based on three alternative approaches for determining the cumulative change in relative sea-level: (i) For all three study sites, the trend in relative sea-level determined from fitting a linear regression model to the mean monthly sea-levels over 55 years from the Pago Pago tide gauge was used; (ii) for Nu'uuli where a linear temporal trend in sediment surface elevation was observed, the rate of change in site specific relative sea-level was used; and (iii) for all three study sites, the

difference between the average of the three mean annual sea-levels around the year of a remotely sensed image and for the average of the three mean annual sea-levels around the year of the first available image (1961 for the three sites) was used. For the third method, for Nu'uuli mangrove, the mean annual sea-levels observed from the tide gauge data were adjusted for the observed linear trend in sediment surface elevation to determine the site specific cumulative change in relative sea-level.

For example, for Nu'uuli mangrove, using the date of the first available aerial photo of 16 September 1961 to the next available imagery of 11 July 1971, an elapsed period of 9.817 years, using the first method, the cumulative change in sea-level was 19.34 mm ( $9.817 \text{ a} \times 1.97 \text{ mma}^{-1}$ ). Using the second method and the same pair of historical images, the calculated rate of site specific relative sea-level rise is  $2.22 \text{ mm a}^{-1}$  and thus the cumulative change in sea-level over this period was 21.79 mm ( $9.817 \text{ a} \times 2.22 \text{ mm a}^{-1}$ ). Using the third method and pair of historical aerial photos, the mean annual sea-levels as measured by the tide gauge for the three years around 1961 (1960, 1961 and 1962) was 79.33 mm, the average for the three years around 1971 was 86.33 mm, an elapsed period of 10 years, and the difference between the two averages is +7.00 mm. The difference in average mean annual site specific sea-levels is +29.00 mm (the observed trend in Nu'uuli sediment surface elevation was  $-0.25 \text{ mm a}^{-1}$ , where  $7.00 \text{ mm} - (-0.25 \text{ mm a}^{-1} \times 10 \text{ a}) = 9.5 \text{ mm}$ ). Advantages/disadvantages of the alternative methods are discussed in Section 5.6.

### **3.7. MANGROVE RESISTANCE TO CHANGES IN RELATIVE SEA-LEVEL**

Observations of past and predicted future trends in site-specific relative sea-level were determined in order to identify each mangrove study site's vulnerability to regional relative sea-level rise. The past rate of site-specific relative sea-level was calculated through comparison of the observed rate of change in mangrove sediment elevation and observed regional relative-sea level rise rate. This is calculated using equation 1, first presented in Section 1.4:

$$h_{rslm} = h_{rsltg} - h_{em} \quad (\text{Equation 1}).$$

As explained in Section 1.4, the rate of change of relative sea-level as measured at a tide gauge may differ substantially from the relative sea-level rate of change occurring in coastal wetlands due to changing elevation of the wetland sediment surface, referred to in this thesis as change in “site specific relative sea-level”. For sites where  $h_{rslm}$ , the site-specific rate of change in relative sea-level, is observed to be positive, this indicates that the mangrove system has not been keeping pace with rising sea-level, and as a result, assuming change in sea-level is the primary control on mangrove position, the mangrove has likely been transgressing landward. If  $h_{rslm} = 0$ , this indicates that the mangrove has not experienced a change in sea-level, and as a result, the mangrove position has likely been stable. A negative value for  $h_{rslm}$  indicates that the mangrove has experienced lowering in sea-level, and as a result, the mangrove has likely been prograding seaward.

Rates of change in site-specific relative sea-level are estimated through the year 2100 for Nu'uuli and Masefau mangroves by applying the global sea-level rise minimum and maximum projections (Church et al., 2001) to determine a range of site-specific relative sea-level projections. When the rate falls within the error interval of IPCC's best estimate of global average sea-level change during the 20<sup>th</sup> century of  $1.5 \pm 0.5 \text{ mm a}^{-1}$ , then the IPCC range of projections for change in global sea-level are applied. Otherwise, the IPCC models providing the minimum and maximum projections for the change in global mean sea level (Appendix II.5, Table II.5.1 in Church et al., 2001) are adjusted by adding a linear rate to the global projection per Equation 2:

Equation 2. Adjusting IPCC models for minimum and maximum projections in global mean sea level rise from 1990-2100 for site-specific relative sea-level rise.

(a) Year 2100 minimum projected site-specific relative mean sea-level  
 $SSRSL_{min} = 92 \text{ mm} + (h_{rslm} - 1.5 \text{ mm a}^{-1}) \times 110 \text{ a}$

(b) Year 2100 maximum projected site-specific relative mean sea level  
 $SSRSL_{max} = 859 \text{ mm} + (h_{rslm} - 1.5 \text{ mm a}^{-1}) \times 110 \text{ a}$

Where  $SSRSL_{min}$  = projected minimum site-specific mean sea-level relative to 1990 mean sea-level.

92 mm = the IPCC minimum projection for rise in global sea level from 1990 to 2100

$h_{rslm}$  = site-specific rate of change in relative sea-level observed at a mangrove site

$1.5 \text{ mm a}^{-1}$  = IPCC's best estimate of global average sea-level change during the 20<sup>th</sup> century

110 a = the elapsed time period from 1990 to 2100 (2100.5 – 1990.5)

$SSRSL_{max}$  = projected maximum site-specific mean sea-level relative to 1990 mean sea-level.

859 mm = the IPCC maximum projection for rise in global sea level from 1990 to 2100

The rate added to the IPCC global projection is the difference between IPCC's best estimate of global average sea level change during the 20<sup>th</sup> century and the observed or estimated rate of change in site specific relative sea-level. For example, if a site specific relative sea-level rise rate ( $h_{rslm}$ ) of  $+4.0 \text{ mm a}^{-1}$  is observed, which is outside of the IPCC estimate of uncertainty for global average sea-level change during the 20<sup>th</sup> century, then  $2.5 \text{ mm a}^{-1}$  ( $4.0 - 1.5$ ) is added to the IPCC models to determine a range of projections for site specific relative sea-level rise at that mangrove site. In this case, at year 2100, 275 mm (from 1990 to 2100,  $110 \text{ a} \times 2.5 \text{ mm a}^{-1}$ ) would be added to the IPCC model projections for change in global mean sea level, resulting



in a range of elevations between 367 and 1,134 mm above mean sea level in 1990. This conceptually translates to linear rates of projected site specific relative sea-level rise of between 3.3 – 10.3 mm a<sup>-1</sup> over the 110 year period, however, in reality the rise will follow a model with an acceleration term and not a linear trend. Equation 2 can be modified for projections from 2004-2100 by using 64 mm and 831 mm for the IPCC minimum and maximum projections for rise in global sea-level, respectively, and using an elapsed time of 96 years (2100.5 – 2004.5).

### **3.8. MANGROVE PHYSIOGRAPHIC SETTING AND RESILIENCE TO RELATIVE SEA-LEVEL RISE**

The mean slope of the land immediately adjacent and upslope to the landward mangrove margins was estimated using a GIS, which included the delineation of 2002 landward mangrove margins by Bardi and Mann (2004) and topography (3.048 m [10 foot] contour interval, American Samoa 1962 datum). Every 50 m along the landward mangrove margin, the distance between the landward mangrove margin and 3.048 m contour was measured. The average of the slopes of the points 50 m along the landward mangrove margin were then determined.

Mangroves are generally located between the level of mean high water spring tides (just above the high tide line) and mean sea level (although there is some variability primarily in the position of the seaward margin), the upper half of the tidal range (Woodroffe, 1992, 2002; Lewis, 2005; Ellison, 2001, 2004, 2006). In American Samoa, the tidal range is 1.07 m, placing the delineated landward mangrove margin roughly at 0.54 m above mean sea-level in 2002, when the mangrove boundary was mapped. Using the observed regional relative sea-level rise rate of 1.97 mm a<sup>-1</sup>, the 3.048 m contour interval based on the 1962 datum, the height datum used in the topographic map, is 3.127 m above the 2002 mean sea level. Equation 3 explains and Fig. 3.5 illustrates the method employed to determine the slope of the land adjacent to landward mangrove margins.

Equation 3. Average slope of the land upslope from the mangrove landward margin.

$$\% \text{ slope} = \alpha = \left[ \frac{(3.127 \text{ m} - \theta_{lm})}{X} \right] \times 100 = \frac{2.587 \text{ m}}{X} \times 100$$

Where  $\alpha$  = Average slope of the land upslope from the mangrove landward margin (average of measurements taken every 50 m along the mangrove boundary)

3.127 m = Constant contour interval, American Samoa 2002 datum [1962 datum 3.048 m contour interval + (1.97 mm a<sup>-1</sup> regional relative sea-level rise rate) x 40 years (2002-1962)]

$\theta_{lm}$  = Estimated elevation of the landward mangrove margin above 2002 msl (0.54 m)

X = Distance between 2002 landward mangrove margin and 3.127 m contour interval 2002 datum average of measurements taken every 50 m along the mangrove landward margin

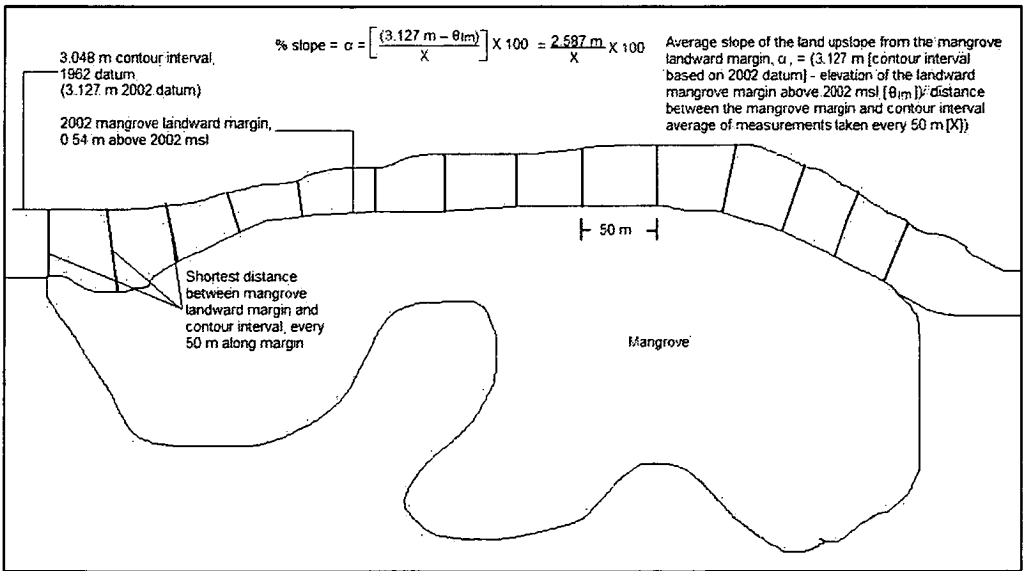


Fig. 3.5. Illustration demonstrating the method employed to estimate the slope of land upslope from the landward mangrove margin.

Estimates of migration of mangrove margins using a predictive model of beach erosion called the Bruun rule (Bruun, 1962, 1988), or a modified Bruun rule, was not employed for this study as mangroves are not expected to respond in accordance with Bruun rule assumptions and because the Bruun rule, as with other general predictive models of coastal erosion, is not suitable for small spatial-scale, site-specific estimates, and over short time periods (Bruun, 1988; SCOR Working Group, 1991; List et al., 1997; Komar, 1998; El-Raey et al., 1999; Pilkey and Cooper, 2004). Section 5.4 provides a detailed discussion of the applicability of Bruun Rule model assumptions to mangroves and use for small temporal and spatial scales.

A GIS is used, including layers for buildings, roads, 2002 landward mangrove margin, and satellite imagery, to identify the location of buildings, seawalls and roads presenting an obstacle to mangrove landward migration through the upper projection for landward transgression of the landward mangrove margin. Information from these layers is used in Section 3.9.2.

### **3.9. PREDICTED CHANGE IN MANGROVE AREA AND POSITION**

#### **3.9.1. Seaward Margin**

Projections for the landward transgression of the seaward mangrove margin were made through the year 2100. These are based on information on the observed mean rate of change in position of the seaward margin and projected site-specific relative sea-level rise scenarios.

For a mangrove site where there is a significant correlation between changes in relative sea-level and position of the seaward margin, a range of projections for the year 2100 seaward mangrove margin position was made by: (i) extrapolating from the observed rate of change in shoreline position into the future, (ii) applying the IPCC lower and (iii) upper global sea-level projections (Church et al., 2001) to each study site's site-specific relative sea-level change rate, and (iv) extrapolating from rates of change in site-specific relative sea-level.

The change in mangrove area resulting from movement of the seaward mangrove margin was conducted by use of a GIS and inclusion of a

fixed line, a GIS shape file with fixed coordinates, in each historical image. The change in mangrove area from two points in time as a result of movement of the mangrove seaward margin is calculated using a GIS by taking the difference between the time 1 (T1) area of open water between the mangrove seaward margin and a fixed line GIS shape file and T2 area of open water. When the area of open water at T2 > area at T1, the average mangrove seaward margin position migrated landward. The resulting observed increase in open water area is equal to the reduction in mangrove area resulting from the net landward mangrove migration of the mangrove seaward margin. Equation 4 and Fig. 3.6 demonstrate this method. Equation 4. Calculating the change in mangrove area resulting from observed movement of the seaward mangrove margin.

$$\Delta \text{ mangrove area from } t_1 \text{ to } t_2 = \text{open water area } t_2 - \text{open water area } t_1$$

The open water area is the area of the polygon with a perimeter comprised of the mangrove seaward margin and a fixed line GIS shapefile.

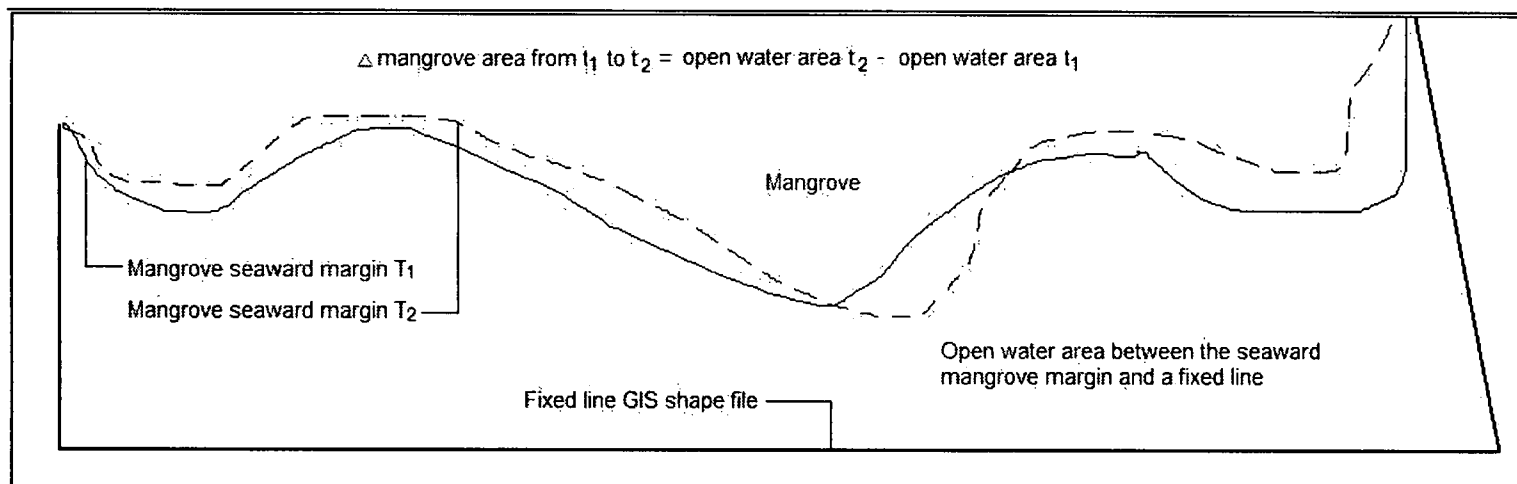


Fig. 3.6. Method for measuring the change in mangrove area resulting from change in position of the seaward mangrove margin.

When there is a significant correlation between the change in site-specific relative sea-level and change in mangrove area resulting from the landward transgression of the seaward mangrove margin for a mangrove site, it is inferred that the ratio of change in mangrove area to change in site-specific relative sea-level over an observed historical time period will be equal to the ratio of the future change in mangrove area to projected change in regional relative sea-level. Using a hypothetical example to demonstrate this method, using the IPCC high projection, Equation 5 is used:

Equation 5. Predicting future change in mangrove area from movement of the mangrove seaward margin based on the observed change in mangrove area and rate of change in site-specific relative sea-level, and projected scenarios for future site-specific relative sea-level.

Observed Past      Predicted Future

$$\frac{\Delta \text{Area } t_1}{\Delta \text{SSRSL } t_1} = \frac{\Delta \text{Area } t_2}{\Delta \text{SSRSL } t_2}$$

such that  $\Delta \text{Area } t_2 = \Delta \text{SSRSL } t_2 \times (\Delta \text{Area } t_1 / \Delta \text{SSRSL } t_1)$

where  $t_1$  = the period from 16 September 1961 to 15 December 2003, and

$t_2$  = the period from 15 December 2003 to 15 June 2100

$\Delta \text{SSRSL}$  = change in site specific relative sea-level over a given time period

Using values for  $\Delta \text{Area } t_1 = 23,160.2 \text{ m}^2$ ,  $\Delta \text{SSRSL } t_1 = 0.0937 \text{ m}$  and projected  $\Delta \text{SSRSL } t_2 = 0.885 \text{ m}$  for the year 2100 from the site specific adjusted IPCC upper projection for global sea level rise, then the predicted change in mangrove area resulting from the landward transgression of the seaward mangrove margin from 15 December 2003 to 15 June 2100 is  $218,749 \text{ m}^2$ . A similar calculation could be made using the site specific adjusted IPCC low projection for global sea level rise through 2100.

Similarly, the method to estimate the rate of migration of the seaward margin is shown in Equation 6:

Equation 6. Predicting the rate of migration of the seaward mangrove margin employing different scenarios for projected change in site specific relative sea-level rise:

Observed Past

Predicted Future

$$\frac{\text{Rate seaward mangrove margin migration } t_1}{\text{Site specific relative sea-level rise } t_1} = \frac{\text{Rate seaward mangrove margin migration } t_2}{\text{Site specific relative sea-level rise } t_2}$$

Such that

$$\text{Rate seaward mangrove margin migration } t_2 = \text{Site specific relative sea-level rise } t_2 \left[ \frac{\text{Rate seaward mangrove margin migration } t_1}{\text{Site specific relative sea-level rise } t_1} \right]$$

Otherwise, for a mangrove site with no significant correlation between site-specific relative sea-level and position of the seaward margin, the year 2100 seaward mangrove margin position was estimated by extrapolating only from the observed rate of change in mangrove area (calculated from the observed change in shoreline position) into the future.

### 3.9.2. Landward Margin

Three alternative scenarios for projected rates of change in relative sea-level were used to estimate the distance that the landward mangrove margin will transgress landward through the year 2100, calculated through Equation 7, and illustrated in Fig. 3.7. The method uses the calculated mean slope of the land immediately adjacent and upslope from the landward mangrove margin and information on the existence of obstacles to mangrove landward migration through the upper projection for landward transgression of the landward mangrove margin (Section 3.8).

Equation 7. Predicting the distance the landward mangrove margin will migrate and concomitant change in mangrove area through the year 2100 based on three scenarios for rates of site specific relative sea-level rise.

$$(a) M = \Delta \text{ RSLR} / \alpha$$

$$(b) \Delta \text{ Area} = (L - B) \times M$$

$$(c) \text{ Rate of } \Delta \text{ Area} = [(L - B) \times M] / 96 \text{ a}$$

$$(d) \text{ Rate of landward margin migration} = M / 96 \text{ a}$$

Where

M = distance landward margin migrates from 2004 through 2100 (96 years)

$\Delta$  RSLR = change in elevation of relative sea-level based on one of three scenarios: (i) IPCC lower projection, (ii) extrapolation from observed relative sea-level rise rate, and (iii) IPCC upper projection

$\alpha$  = slope (Section 3.8, Equation 3)

$\Delta$  Area = change in mangrove area from 2004 through 2100 resulting from the movement of the landward margin

L = most current length of the landward margin

B = length of the landward margin that is obstructed from landward migration

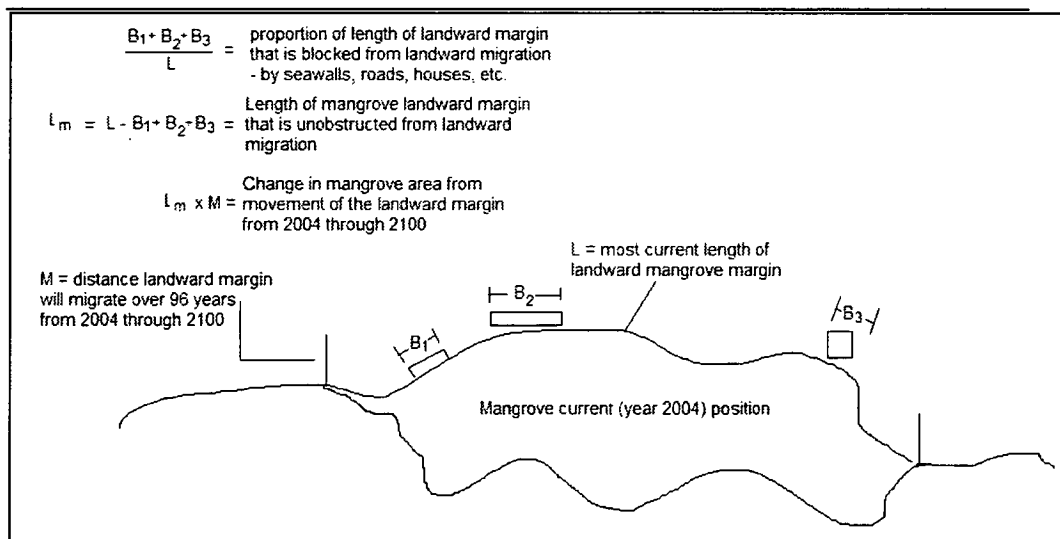


Fig. 3.7. Method to predict change in mangrove area resulting from movement of the landward margin.

The estimate used the 9 November 2003 and 15 December 2003 lengths of the Masefau and Nu'uuli mangrove landward margins of 1,272 m and 6,423 m, respectively, and the 20 May 2004 length of the Leone mangrove



landward margin of 1,273 m, reduced by the length of the margin that is obstructed from migrating landward.

Assumptions in conducting this analysis are that the landward mangrove margin is located at just above the mean high tide line, and has not been altered by human activities, such as filling and placement of seawalls; the sediment composition of the upland habitat where the mangrove might migrate is suitable for mangrove establishment; the slope of land upslope from the mangrove landward margin will not change in response to site-specific sea-level rise; there will be no new obstacles to landward migration of the landward mangrove margins between now and the year 2100; and there is no lateral movement of the mangrove.

### **3.10. REGIONAL EXTRAPOLATION**

Projected changes in mangrove area through the year 2100 for countries and territories in the Pacific Islands region with indigenous mangroves were made by extrapolating from results from American Samoa on projected rates of change in mangrove sediment elevation, regional relative sea-level rise, and mangrove area. Hawaii and Tahiti, where mangrove were introduced (Ellison, 1999; Allen 1998; Chimner et al., 2006), are not included in the assessment because management authorities in these areas may actively control the alien invasive species (e.g., Smith, 2005). While mangrove tree species have been reported from Niue (Ellison, 1999) where one true mangrove species *Excoecaria agallocha* is documented (Yuncker, 1943; Tomlinson, 1986), this species is also found in non-wetland habitat such as dry littoral forest, and a national government contact reported that there are no mangrove wetlands in Niue (personal communication, 10 June 2005, Fiafia Rex, Fisheries Division, Niue Department of Agriculture Forestry and Fisheries).

For the purpose of this regional extrapolation, if an estimated rate of change in site-specific relative sea-level is  $\leq 0$  (sea-level is stable or lowering), then it is assumed that the mangrove area will remain stable through 2100. For sites where the rate of change in site-specific relative sea-

level is positive, Equation 8 is used to provide a rough estimate of change in mangrove area:

Equation 8. Rough estimate of change in mangrove area by the year 2100 for Pacific Island Countries and territories with indigenous mangroves.

$$\frac{\% \Delta \text{Area}_{\text{AS}}}{h_{\text{rslmAS}}} = \frac{\% \Delta \text{Area}_{\text{PIC}}}{h_{\text{rslPIC}}}$$

Where  $\% \Delta \text{Area}_{\text{AS}}$  is the estimated percent change in mangrove area for American Samoa through 2100,  $h_{\text{rslmAS}}$  is the average American Samoa mangrove site-specific relative sea-level rise rate,  $\% \Delta \text{Area}_{\text{PIC}}$  is the estimated percent change in mangrove area of a specific Pacific Island Country or territory, and  $h_{\text{rslPIC}}$  is the estimated mangrove site-specific relative sea-level rise rate of that country or territory.

When extrapolating sea-level to the year 2100 from the observed or estimated rate of mangrove site-specific relative sea-level rise,  $h_{\text{rslPIC}}$  = the observed regional relative sea-level change rate minus the rate of change in elevation of the mangrove sediment surface. When applying the IPCC upper projection for global sea-level rise through 2100, Equation 2(b) is employed, which uses adjusted IPCC models A1T and A1FI, which provide the maximum projections for the change in global mean sea level (Appendix II.5, Table II.5.1 in Church et al., 2001).

When extrapolating from the observed American Samoa mangrove site-specific relative sea-level rise rate through the year 2100, the average of rates for Nu'uuli and Masefau mangroves is used for  $h_{\text{rslmAS}}$ . The average site specific relative sea-level rise rate for American Samoa is  $2.10 \text{ mm a}^{-1}$  when extrapolating from the historical record (Nu'uuli  $2.22 \text{ mm a}^{-1}$ , Masefau  $1.97 \text{ mm a}^{-1}$ , Section 4.7). The average rate is  $9.17 \text{ mm a}^{-1}$  when applying the adjusted IPCC upper scenario. We employ the unadjusted IPCC projection for Masefau, a rise of 831 mm over 96 years or  $8.66 \text{ mm a}^{-1}$ , because at this site the site specific sea-level rise rate falls within the IPCC estimate of global average sea-level change during the 20<sup>th</sup> century (Church et al., 2001; Section 4.7. For Nu'uuli, we use equation 2(b) to adjust the rate

by  $+0.72 \text{ mm a}^{-1}$  ( $2.22 \text{ mm a}^{-1} - 1.5 \text{ mm a}^{-1}$ , where 2.22 is the Nu'uuli observed site specific relative sea-level rise rate, and 1.5 is IPCC's best estimate of global average sea-level change during the 20<sup>th</sup> century). Leone mangrove projections are not included in the calculation of  $h_{\text{rslmAS}}$  because it was determined that factors other than change in relative sea-level are primary controls on change in position and area. The estimated percent change in area of Nu'uuli, Masefau and Leone mangroves is used for  $\% \Delta \text{Area}_{\text{AS}}$  (Section 4.8).

These calculations assume that the slope of the land adjacent and upslope from the mangroves, presence of obstacles to landward migration of mangroves, ratio of total mangrove area to length of seaward and landward margins, and other factors are similar at these sites as observed at the American Samoa mangrove sites. Also, regional relative sea-level trends may be based on analysis of data from a tide gauge located far from the mangrove sites and may not reflect the relative sea-level trend at the mangrove locations.

For areas lacking data on mangrove sediment elevation rates, we make the rough estimate that low island mangroves (low limestone or organic origin) have a  $+0.5 \text{ mm a}^{-1}$  rate of rise in sediment elevation, and high island mangroves (on islands that are volcanic or volcanic/raised limestone assemblages) the rate is estimated to be  $+1.5 \text{ mm a}^{-1}$ . Information on rates of change in mangrove sediment surface elevation are available for American Samoa from this study (average of  $-1.45 \text{ mm a}^{-1}$ ), the Federated States of Micronesia (average of  $+1.3 \text{ mm a}^{-1}$ ) (Krauss et al., 2003), and 28 sites located in the Wider Caribbean Region and western and central Pacific (average of  $+1.0 \text{ mm a}^{-1}$ ) (Cahoon and Hensel, 2006). Ellison and Stoddart (1991) document sediment accretion rates of about  $1.2 \text{ mm a}^{-1}$  on low Pacific Islands and  $4.5 \text{ mm a}^{-1}$  on high islands.

Regional relative sea-level change rates for the countries and territories with indigenous mangroves are calculated from fitting a linear regression model to mean monthly relative sea-levels from historical tide gauge records with tidal constituents removed for Pohnpei, Federated States of Micronesia; Suva, Fiji; Guam, USA; Kanton Island, Republic of Kiribati; Majuro, Republic of the Marshall Islands; Noumea, New Caledonia, France;

Malakal, Palau; Honiara, Solomon Islands; and Funafuti, Tuvalu. Rates are calculated from fitting a linear regression model to mean monthly relative sea-levels for Pago Pago, American Samoa; Nauru; and Port Moresby, Papua New Guinea. For Saipan, Commonwealth of the Northern Mariana Islands, USA; Apia, Samoa; Nuku'alofa, Tonga; and Port Vila, Vanuatu, sites with a local tide gauge record of < 20 years, relative sea-level change trends are calculated from TOPEX/Poseidon satellite altimetry data combined with historical global tide gauge records over the period 1950-2001 employing the method by Church et al. (2004a). Papua New Guinea has less than 20-year tide gauge record but results from reconstructed analysis were not available.

Chapter 4

Results

American Samoa Mangrove Vulnerability and Responses to  
Relative Sea-Level Rise

4.1. PROJECTED TRENDS IN AMERICAN SAMOA RELATIVE SEA-  
LEVEL

Fig. 4.1 presents mean monthly relative sea-levels from October 1948 through May 2004 for Pago Pago, American Samoa. Gaps appear in the plot where there were not enough data to produce a reliable monthly mean. A linear regression model was fit to the data series, indicating a mean relative sea-level rise of 1.97 mm a<sup>-1</sup> (± 0.32 95% CI, N = 619) over the observed 54.67 years. Based on the linear regression model, mean sea-level in American Samoa will rise 189 mm between 2004 and 2100. Fig. 4.2 shows the IPCC projections for American Samoa range between 92 and 859 mm rise in relative mean sea-level from 1990 to 2100, and a rise of between 64 mm and 831 mm between 2004 and 2100 (Church et al., 2001).

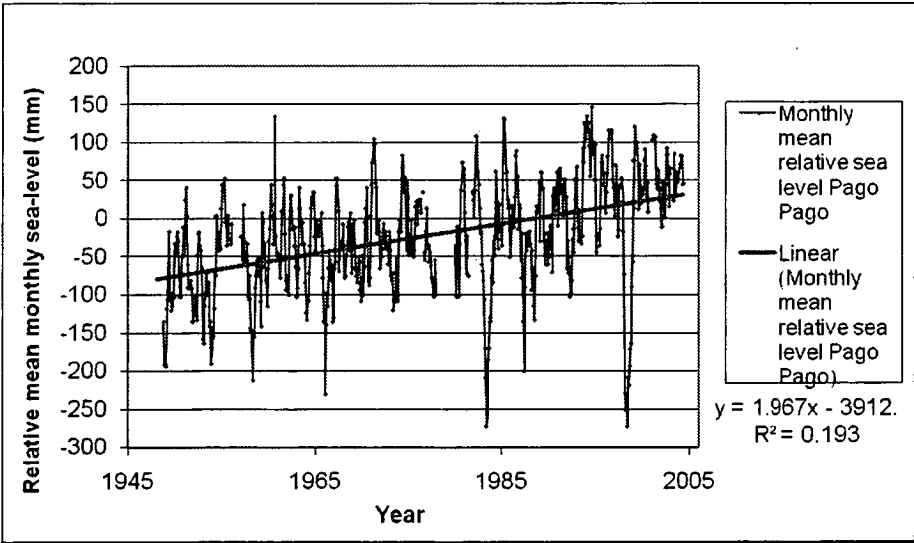


Fig. 4.1. Monthly mean relative sea level from a tide gauge located in Pago Pago harbor, American Samoa, from October 1948 – May 2004.

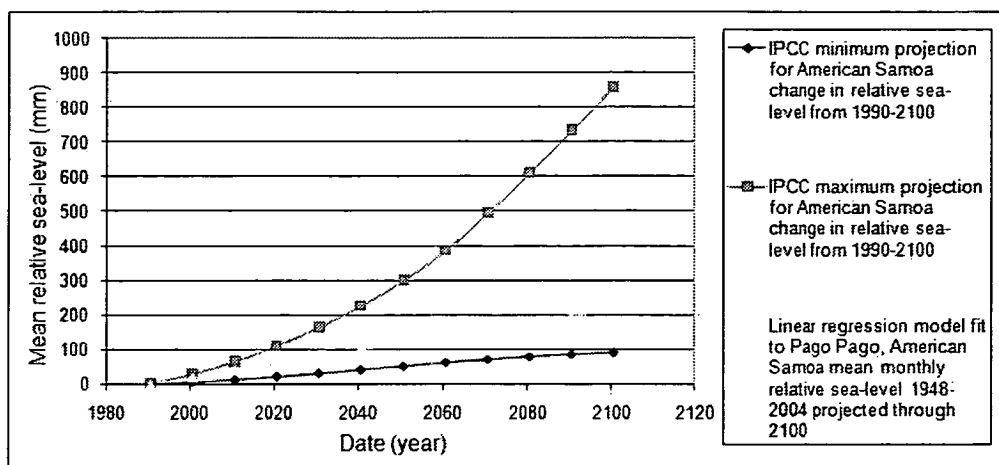


Fig. 4.2. Plot of a linear regression model fit to mean monthly relative sea-levels from tide gauge data for Pago Pago, American Samoa from 1948-2004 extrapolated through 2100, and plots of the IPCC upper and lower projections for the change in mean global sea-level from 1990 through 2100 (Church et al., 2001).

## 4.2. TRENDS IN FREQUENCY AND ELEVATION OF EXTREME HIGH WATER EVENTS AND CAUSES

### 4.2.1. Trends in Frequency of Extreme High Water Events

The 99.95-percentile reduced to median sea-level for the Pago Pago tide gauge data series for the 55 year period is 686.25 mm. This elevation is used to define the minimum value for an extreme high water event at the 99.95 percentile. The 99.9-percentile reduced to median sea-level is 669 mm.

There was zero slope in the trend in occurrence of 99.95-percentile extreme high water events ( $0.00 \text{ events a}^{-1} (\pm 0.019 \text{ 95\% CI, } N = 55)$ ) over the observed 55-year period (Figs. 4.3a and b). There were between 0 and 4 extreme high water events per year, with an average of  $1.5 \text{ events a}^{-1}$ . There were a total of 39 extreme high water events over the first 27 years with an average of  $1.4 \text{ events a}^{-1}$ , and a total of 46 extreme high water events over the latter 27 years, an average of  $1.7 \text{ events a}^{-1}$ .

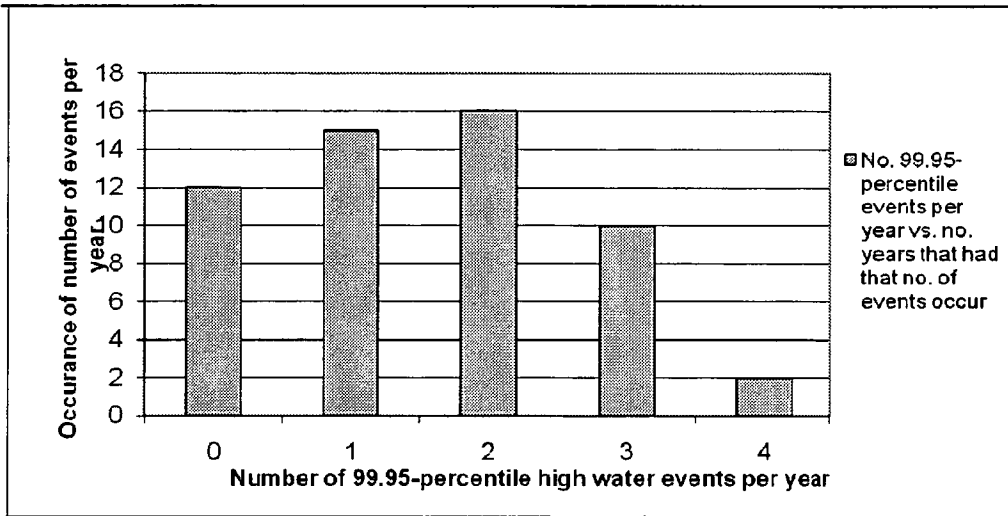
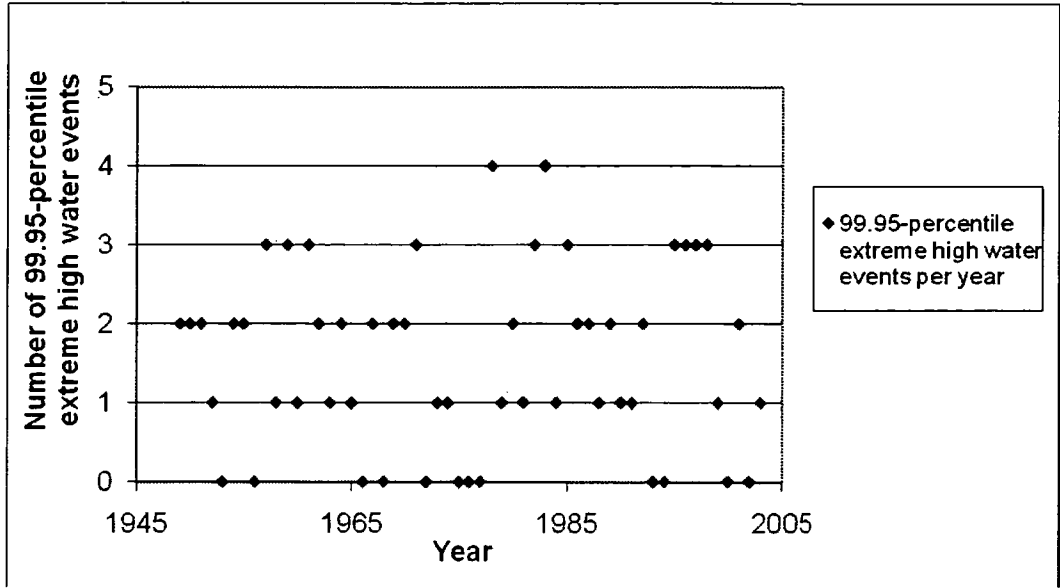


Fig. 4.3a (top). Number of 99.95-percentile extreme high water events (when relative sea-level is  $\geq 686.25$  mm above that year's median sea-level occurring within a single one-week period) per year.

Fig. 4.3b (bottom). Number of times there were the specified number of 99.95-percentile extreme high water events per year, Pago Pago, American Samoa, 1949 - 2003.

There was a  $0.9 \text{ days a}^{-1}$  ( $\pm 3.9$  95% CI) trend in length of return period for 99.95-percentile extreme high water events from 1949-2003 (Fig. 4.4). The probability value from the F-test for the linear regression indicates there is a 96% probability that the coefficients are equal to zero, and the 95%

CI includes 0 indicating no linear trend. The average return time for 99.95-percentile extreme high water events from 1949-2003 was 236.2 days. The average return time decreased from the first to second half of the period from 245.7 to 228.8 days, a decrease in return period by a factor of 0.9 (7% decrease).

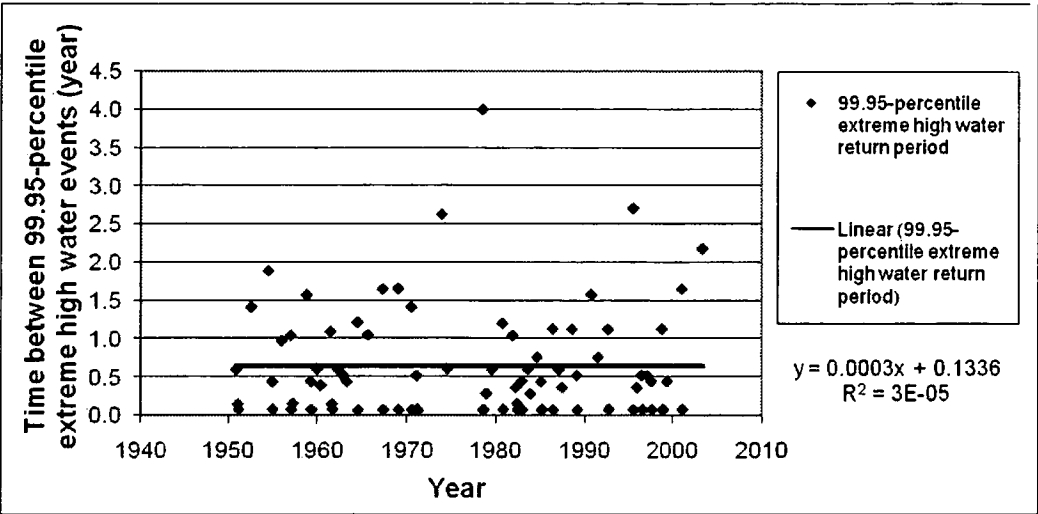


Fig. 4.4. Plot of date versus return time of 99.95-percentile extreme high water events (when relative sea-level is  $\geq 686.25$  mm above that year's median sea-level), Pago Pago, American Samoa, 1949-2003.

There was a trend in occurrence of 99.9-percentile extreme high water events of  $-0.01$  events  $a^{-1}$  ( $\pm 0.02$  95% CI,  $N = 55$ ) (Fig. 4.5a,b). The probability value from the F-test for the linear regression indicates there is 24% probability that the coefficients are equal to zero, and the 95% CI includes 0 indicating no linear trend. There were between 0 and 4 extreme high water events  $a^{-1}$ , with an average of 2.2 events  $a^{-1}$ . There were a total of 65 extreme high water events over the first 27 years with an average of 2.4 events  $a^{-1}$ , and a total of 56 extreme high water events over the latter 27 years, an average of 2.0 events  $a^{-1}$ .



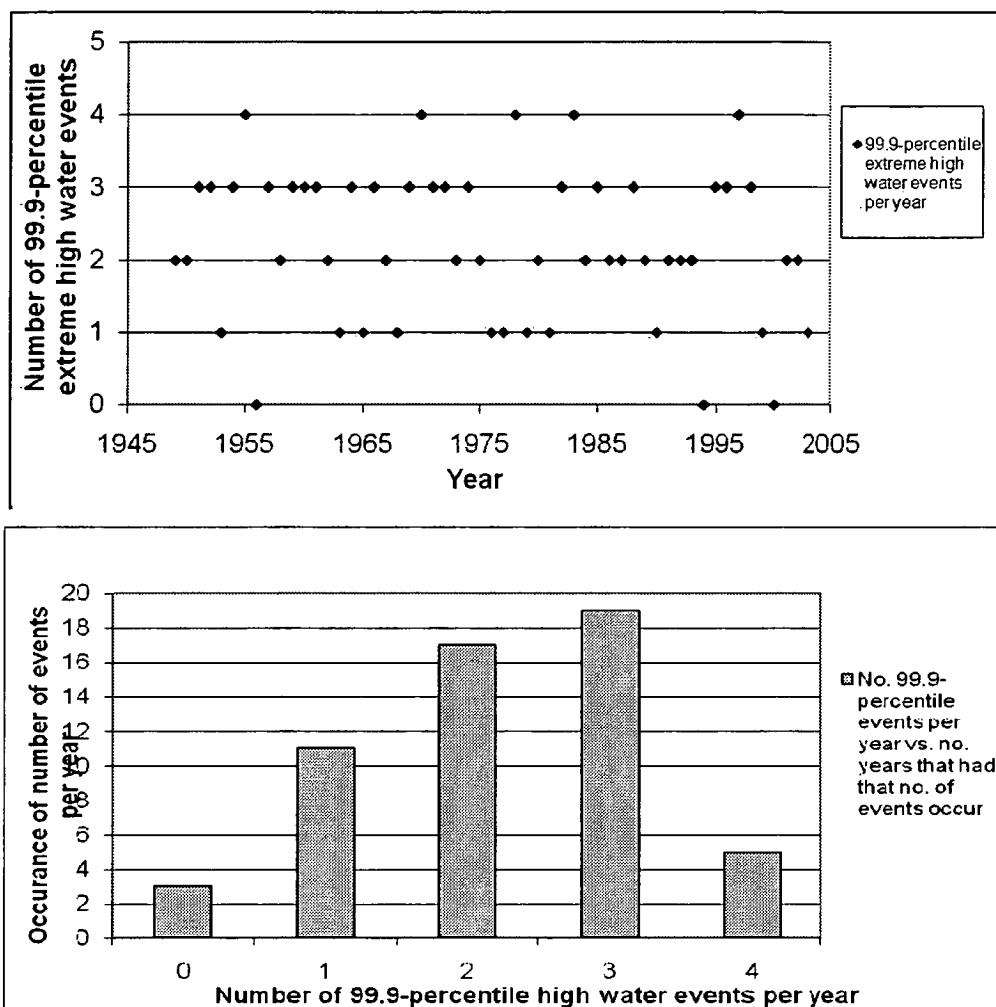


Fig. 4.5a (top). Number of 99.9-percentile extreme high water events (when relative sea-level is  $\geq 669$  mm above that year's median sea-level occurring within a single one-week period) per year.

Fig. 4.5b (bottom). Number of times there were the specified number of 99.9-percentile extreme high water events per year, Pago Pago, American Samoa, 1949 - 2003.

There was a  $0.8 \text{ days a}^{-1} (\pm 1.7 \text{ 95\% CI})$  trend in length of return period for 99.9-percentile extreme high water events from 1949-2003 (Fig. 4.6). The probability value from the F-test for the linear regression indicates there is a 34% probability that the coefficients are equal to zero, and the 95% CI includes 0 or no trend. The average return time for 99.9-percentile extreme high water events from 1949-2003 was 163.0 days. The average

return time increased from the first to second half of the period from 151.1 to 176.4 days, an increase in return period by a factor of 1.2 (17% increase).

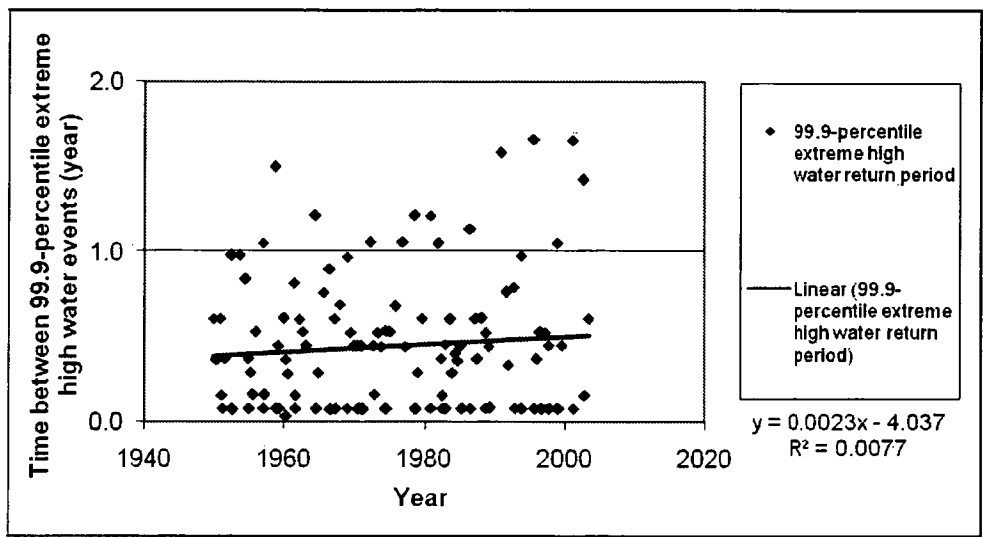


Fig. 4.6. Plot of date versus return time of 99.9-percentile extreme high water events (when relative sea-level is  $\geq 669$  mm above that year’s median sea-level), Pago Pago, American Samoa, 1949-2003.

Fig. 4.7 shows the mean return time of the 99.95, 99.9 as well as 99.8-percentile extreme high water levels for Pago Pago, comparing the two halves of the tide gauge data series from pre and post 15 June 1976. Average return time of 99.95-percentile extreme high water events decreased 7% from the first to second half of the observed 55-year period while the average return time of 99.9-percentile extreme high water events increased 17% (Fig. 4.7). These observed changes in length of return period are both small, where the difference in results for the two different percentiles could be a result of the definition employed of an extreme high water event, which resulted in small data series (N = 81 and 121 for the 99.95 and 99.9-percentile return period analyses, respectively), such that the trends in frequency (number of events per year) of both 99.9 and 99.95-percentile extreme high water events both had confidence intervals that overlapped 0, indicating no linear trend (Fig. 4.4 and 4.6).

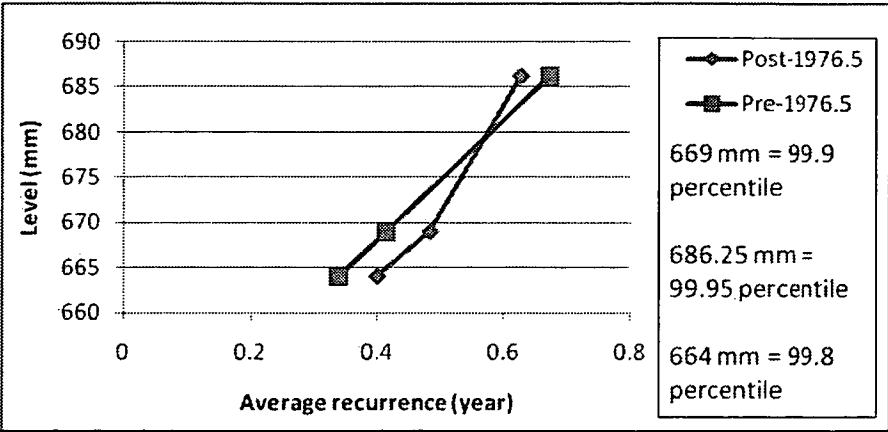


Fig. 4.7. Change in average return time for Pago Pago, American Samoa extreme high water levels 1949 – mid 1976 and mid 1976 – 2003.

#### 4.2.2. Trends in Extreme High Water Event Levels

Figs. 4.8 – 4.9 show that the highest annual 0.1% hourly relative sea-levels have been increasing at  $1.85 \text{ mm a}^{-1}$  ( $\pm 0.29$  95% CI,  $N = 555$ ), the highest annual 0.05% hourly relative sea-levels have been increasing at  $1.85 \text{ mm a}^{-1}$  ( $\pm 0.43$  95% CI,  $N = 281$ ), and annual median hourly sea-levels have been increasing at  $1.71 \text{ mm a}^{-1}$  ( $\pm 0.80$  95% CI,  $N=55$ ) over the observed 55-year period. The trend in monthly mean relative sea-levels is  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI,  $N = 619$ , October 1948 – May 2004) and trend in mean annual relative sea-levels is  $1.92 \text{ mm a}^{-1}$  ( $\pm 0.9$  95% CI,  $N = 49$ , 1949-2003). There are overlapping confidence intervals around the estimates for the trends in extreme high water event levels and trends in relative mean and median sea-levels.

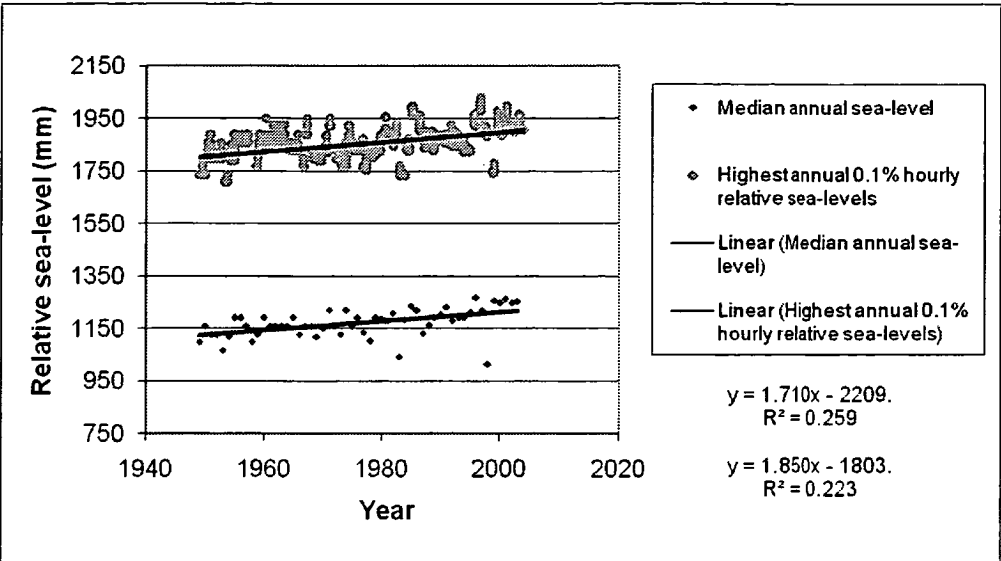


Fig. 4.8. Plot of date versus highest annual 0.1% hourly relative sea-levels, Pago Pago, American Samoa, and plot of annual median sea-level, 1949- 2003.

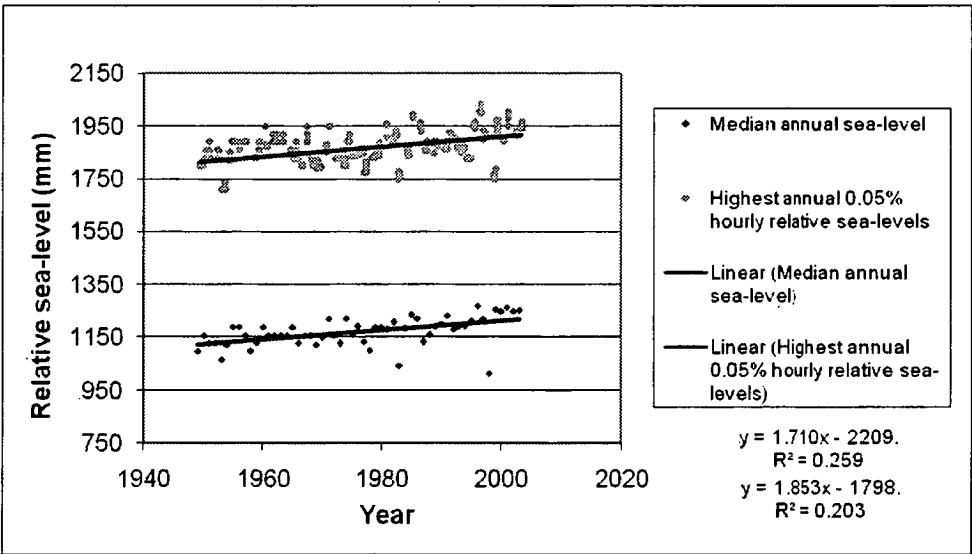


Fig. 4.9. Plot of date versus highest annual 0.05% hourly relative sea-levels, Pago Pago, American Samoa, and plot of annual median sea-level, 1949- 2003.

Fig. 4.10 shows that the highest annual 0.1% hourly relative sea-levels reduced to median sea-level have been increasing at a rate of  $0.11 \text{ mm a}^{-1}$  ( $\pm 0.19$  95% CI,  $N = 555$ ). The probability value from the F-test for the linear

regression indicates there is 27% probability that the coefficients are equal to zero, and the 95% CI includes 0 indicating no linear trend.

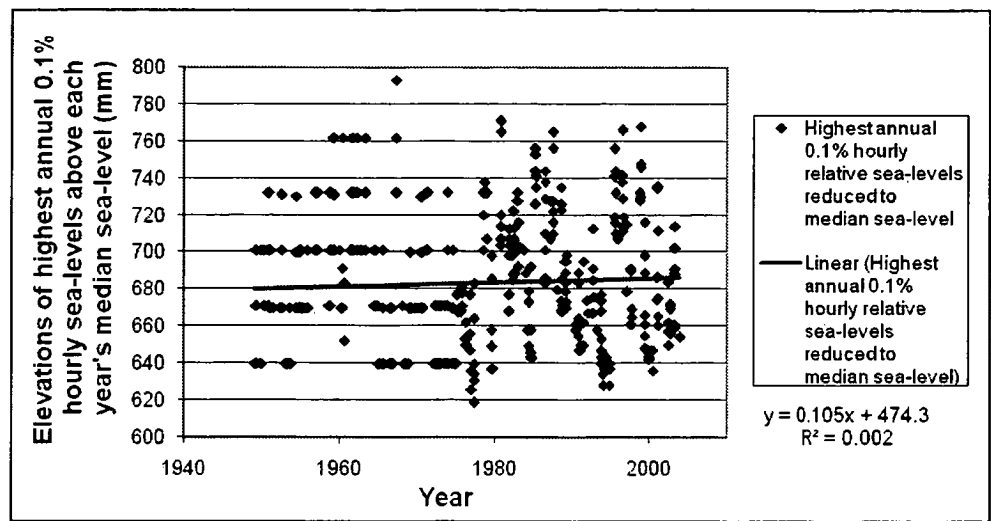


Fig. 4.10. Plot of date versus elevations of the highest annual 0.1% hourly relative sea-levels above each year's median sea-level, Pago Pago, American Samoa, 1949-2003.

Fig. 4.11 shows that the highest 0.05% hourly relative sea-levels reduced to median sea-level have been increasing at a rate of 0.14 mm a<sup>-1</sup> (± 0.28 95% CI, N = 281) over this time period. The probability value from the F-test for the linear regression indicates there is 30% probability that the coefficients are equal to zero, and the 95% CI includes 0 indicating no linear trend.

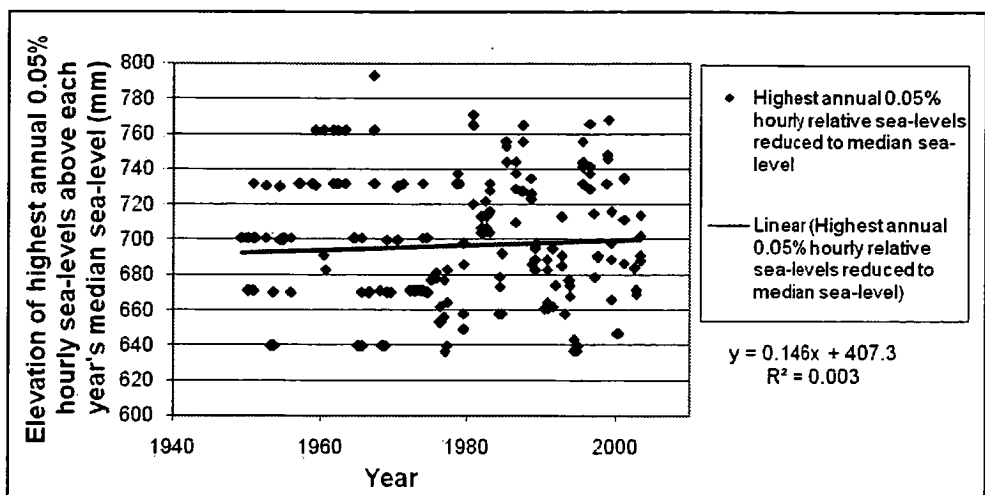


Fig. 4.11. Plot of date versus elevations of the highest annual 0.05% hourly relative sea-levels above each year's median sea-level, Pago Pago, American Samoa, 1949-2003.

#### 4.3. HISTORICAL RECONSTRUCTION OF MANGROVE MARGIN POSITION

Figs.4.12 – 4.14 present the observed change in position of the mangrove seaward margins. Table 4.1 presents the change in mangrove area resulting from movement of the seaward mangrove margin and change in relative sea-level over the four decades for the three mangrove study sites. Fig. 4.15 presents a plot of the cumulative change in mangrove areas with a linear regression model fit to each data series.

# **Masefau time series of mangrove seaward margins overlayed on 1961 imagery**

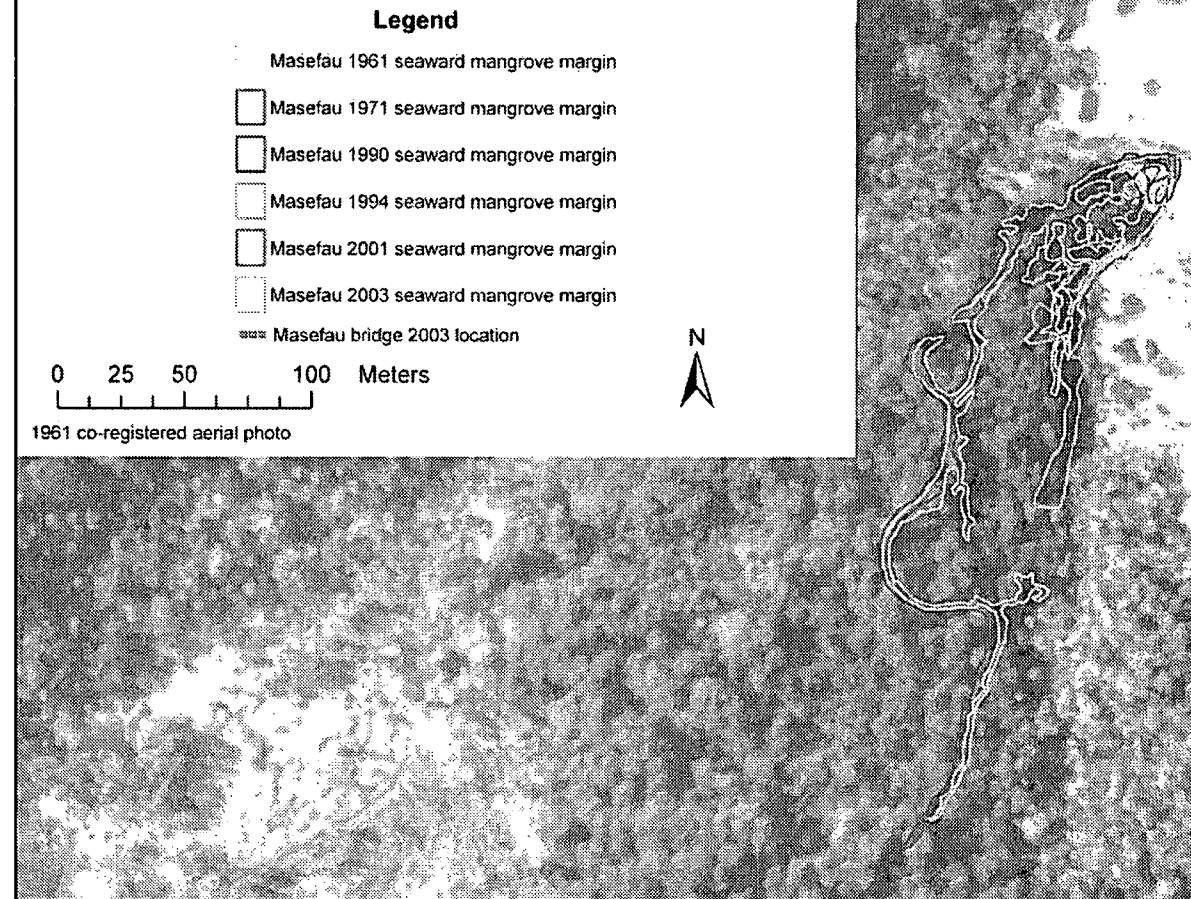


Fig. 4.12a. Time series of Masefau mangrove seaward margin at six points in time from 1961 – 2003 overlaid on a 1961 co-registered aerial photo.

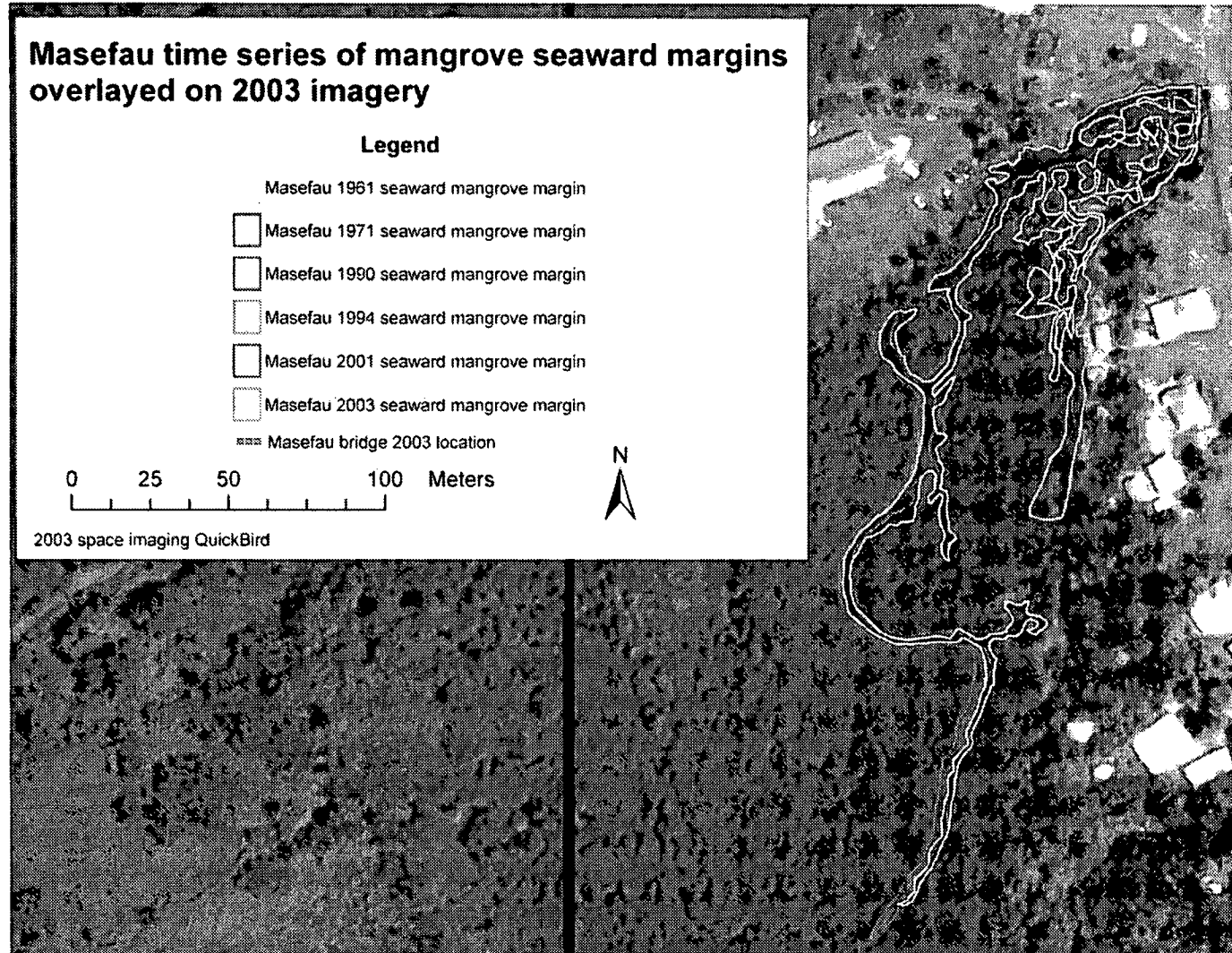


Fig. 4.12b. Time series of Masefau mangrove seaward margin at six points in time from 1961 – 2003 overlaid on a 2003 QuickBird satellite image.



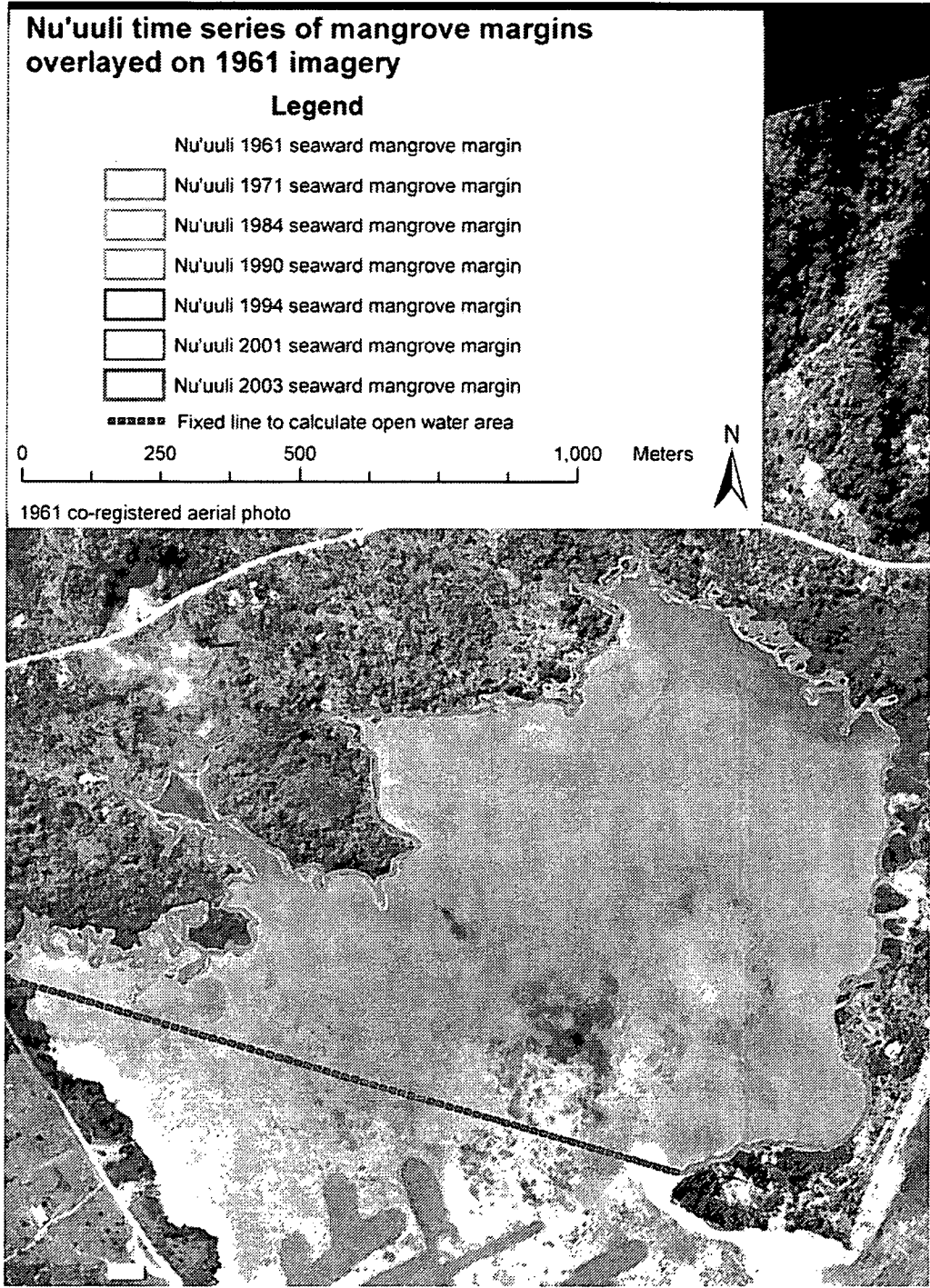


Fig. 4.13a. Time series of Nu'uuli mangrove seaward margin at seven points in time from 1961 – 2003 overlaid on a 1961 co-registered aerial photo.

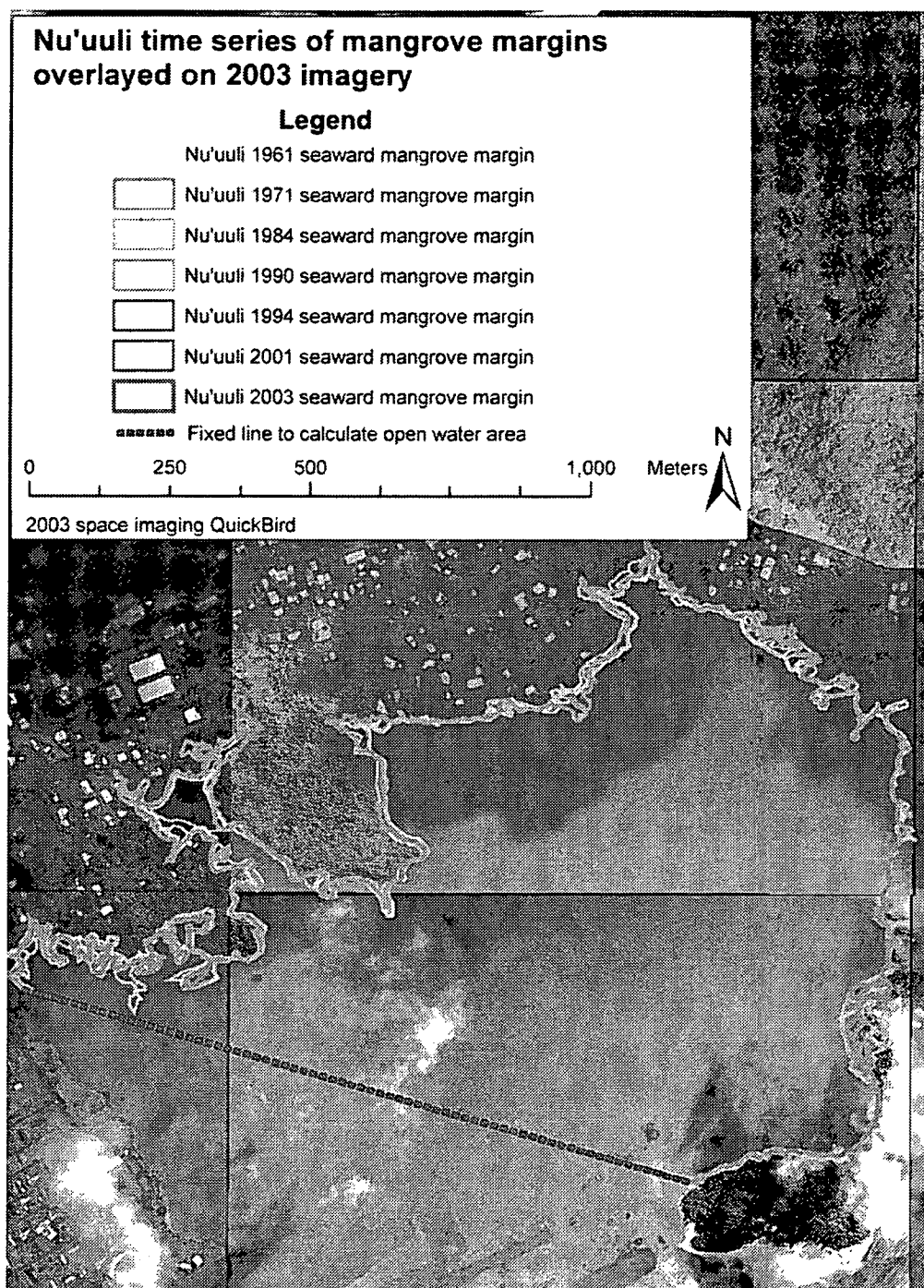


Fig. 4.13b. Time series of Nu'uuli mangrove seaward margin at seven points in time from 1961 – 2003 overlaid on a 2003 QuickBird satellite image.

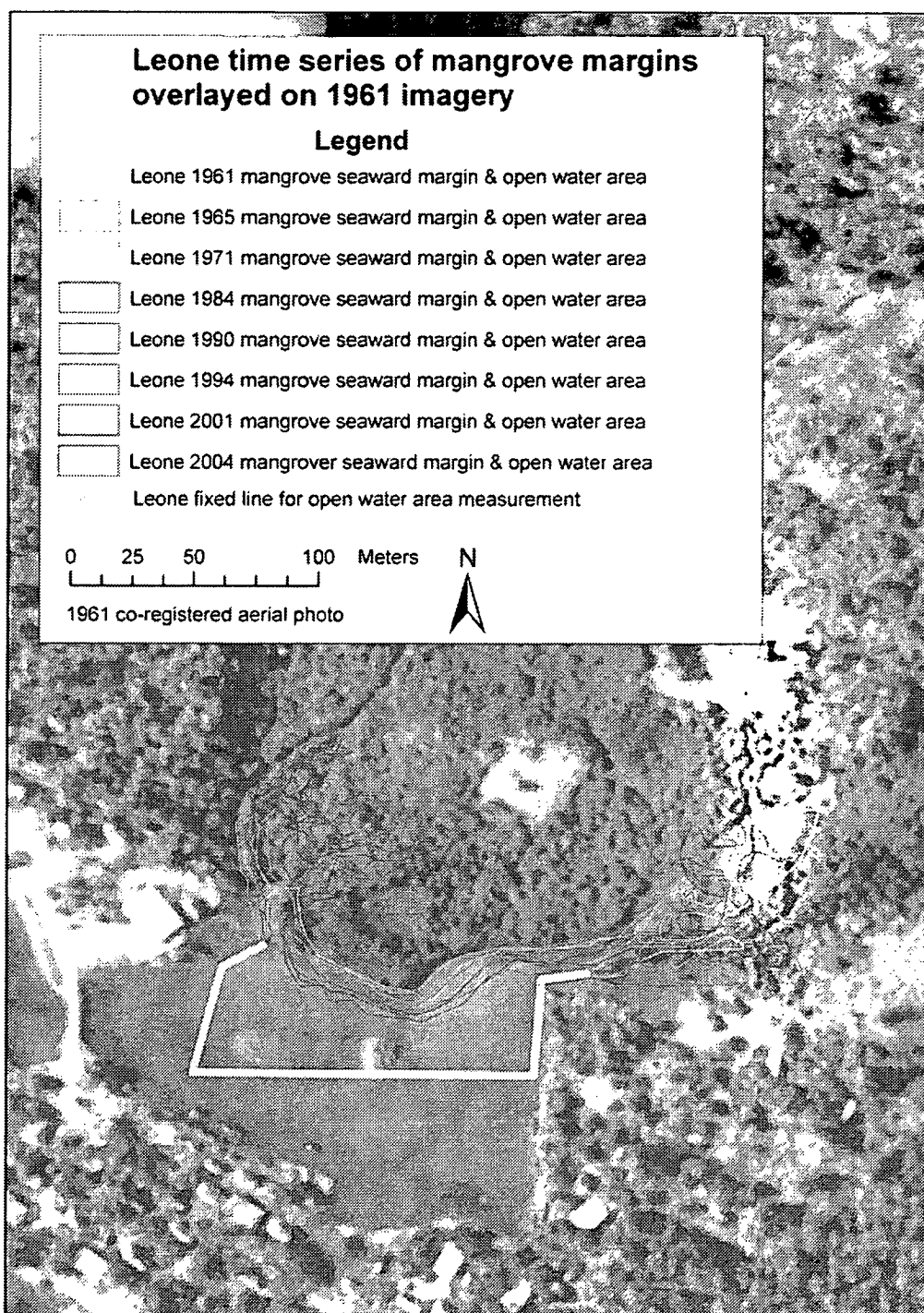


Fig. 4.14a. Time series of Leone mangrove seaward margin at eight points in time from 1961 – 2004 overlaid on a 1961 co-registered aerial photo.

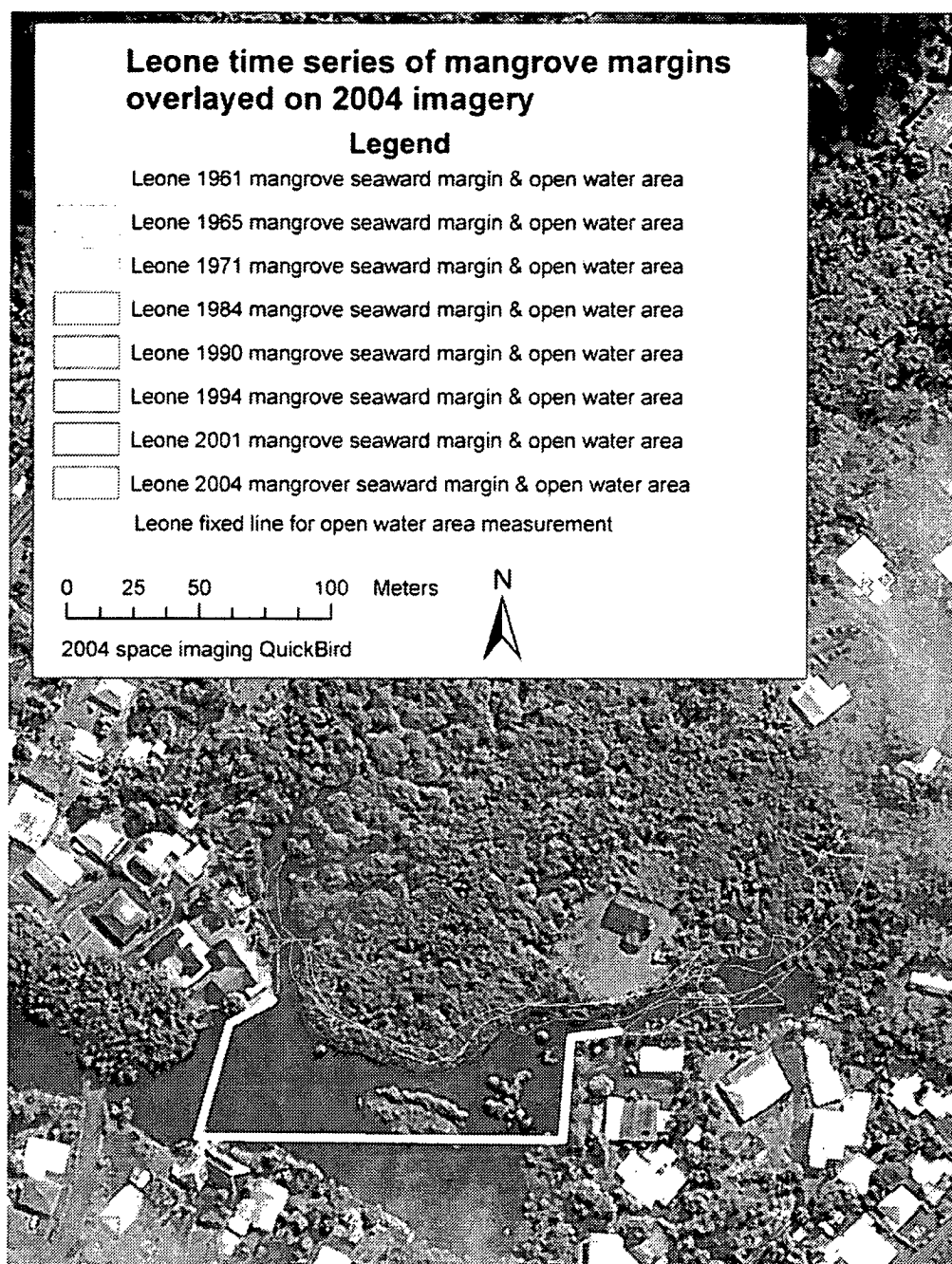


Fig. 4.14b. Time series of Leone mangrove seaward margin at eight points in time from 1961 – 2004 overlaid on a 2004 QuickBird satellite image.

Table 4.1. Change in mangrove area resulting from movement of the seaward mangrove margin and change in relative sea-level, for Masefau, Nu'uuli, and Leone mangroves, American Samoa, 1961-2003/4.

Date of Image (month/day/ year)	Cumulative change in mangrove area from movement of the seaward mangrove margin (m <sup>2</sup> ) <sup>a</sup>	Full 55-year data series cumulative change in relative monthly mean sea-level (mm) <sup>b</sup>	Cumulative change in relative annual mean sea- level (time intervals corresponding to imagery) (mm) <sup>c</sup>
<b>Masefau</b>			
9/16/1961	0	0	0
7/11/1971	-441.37 ± 16	19.34	7.00
9/18/1990	-781.22 ± 100	57.14	26.67
6/30/1994	-1210.47 ± 4	64.59	70.67
9/15/2001	-1901.97	78.80	79.33
11/9/2003	-2203.43	83.03	75.67
<b>Nu'uuli</b>			
9/16/1961	0	0	0
7/11/1971	-8,950.91 ± 9	21.79	9.5
11/7/1984	-13,818.43 ± 4	51.38	-17.58
8/27/1990	-14,140.16 ± 1	64.28	33.92
6/17/1994	-18,312.63 ± 16	72.71	78.92
9/15/2001	-19,719.88	88.80	89.33
12/15/2003	-23,160.22	93.57	86.17
<b>Leone</b>			
9/16/1961	0	0	0
3/31/1966	937.40 ± 16	8.94	-32.67
8/17/1971	984.63 ± 9	19.54	7.00
11/7/1984	-2,150.33 ± 9	45.60	-23.33
9/27/1990	-3,466.10 ± 4	57.19	26.67
6/30/1994	-53.01 ± 9	64.59	70.67
9/15/2001	652.65	78.80	79.33
5/20/2004	198.67	84.07	78.67

<sup>a</sup> Confidence intervals are calculated from the co-registration error from the IKONOS 2001 reference image, reported in Appendix 2.

<sup>b</sup> For Leone mangrove, we use 1.97 mm a<sup>-1</sup>, which is the relative sea-level rise trend from fitting a linear regression model to mean monthly relative sea-levels observed from the Pago Pago, American Samoa tide gauge, October 1948 – May 2004 (Section 3.6, method 1). For Nu'uuli, we use site specific relative sea-level rise (Section 3.6, method 2) (change in sea-level relative to the mangrove sediment surface) of 2.22 mm a<sup>-1</sup> (Section 4.7), and for Masefau we use site specific relative sea-level rise of 1.97 mm a<sup>-1</sup> (Sections 4.4 and 4.7).

<sup>c</sup> Based on annual average mean sea-levels, average of three years around the date of the imagery, adjusted for linear trends in sediment surface elevation for Nu'uuli (Section 3.6, method 3).

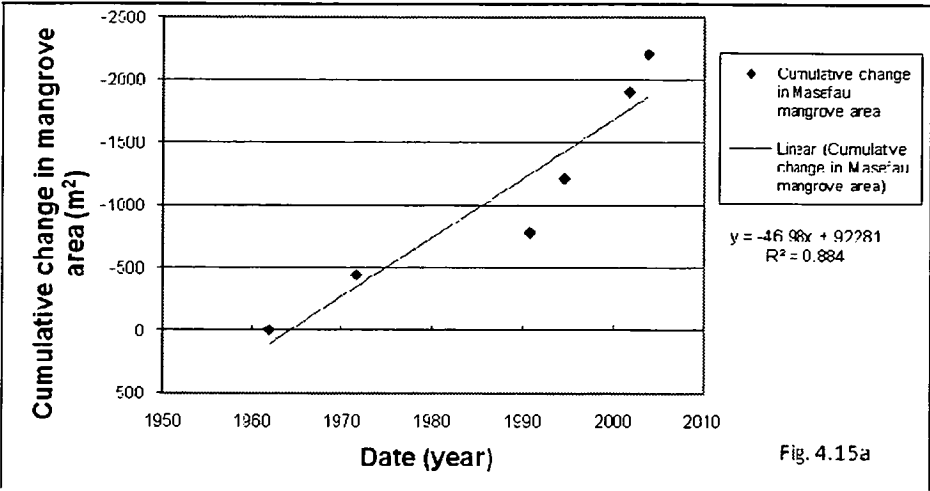


Fig. 4.15a

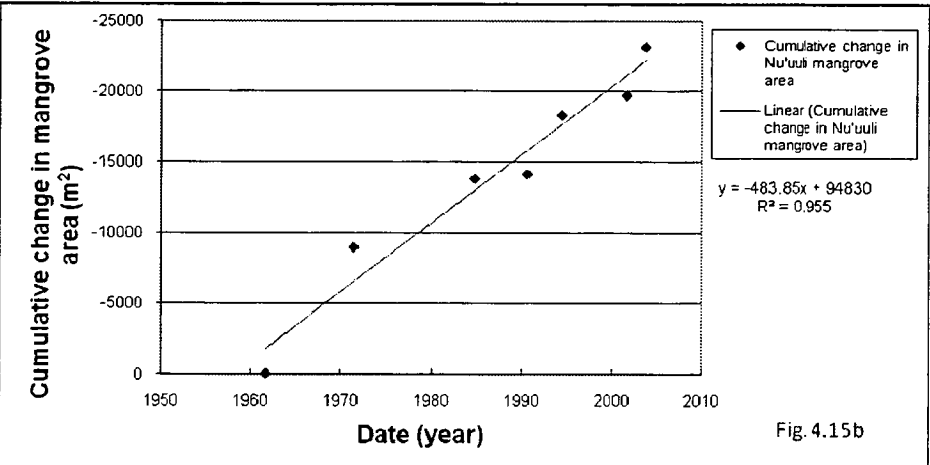


Fig. 4.15b

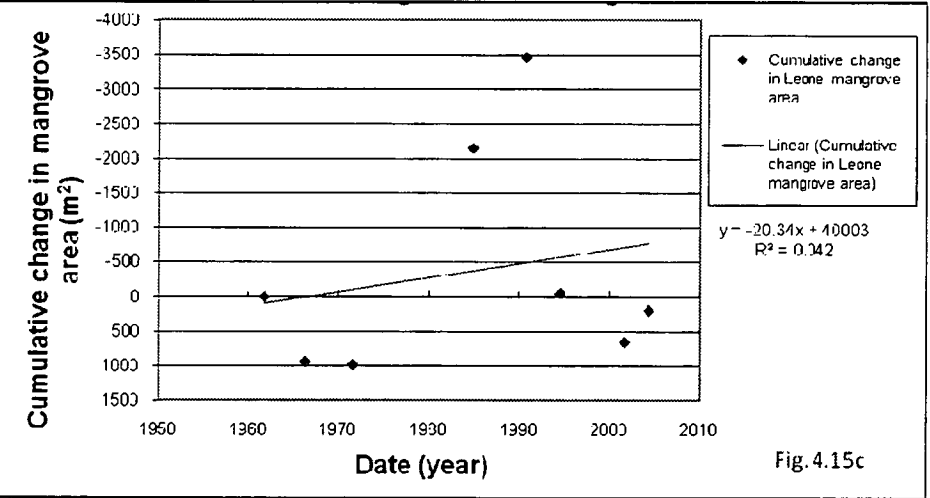


Fig. 4.15c

Fig. 4.15. Plot of the change in mangrove area resulting from movement of the seaward mangrove margin versus date and linear regression model for (a) Masefau, (b) Nu'uuli and (c) Leone mangroves, American Samoa from 1961-2003/4.

The trend in change in Masefau's mangrove area resulting from movement of the seaward mangrove margin, based on linear regression analysis, is  $-46.98 \text{ m}^2 \text{ a}^{-1}$  ( $\pm 23.6$  95% CI,  $R^2 = 0.88$ ,  $P < 0.01$ ). Nu'uuli's trend is  $-483.85 \text{ m}^2 \text{ a}^{-1}$  ( $\pm 119.7$  95% CI,  $R^2 = 0.96$ ,  $P < 0.01$ ). Leone's trend is  $-20.34 \text{ m}^2 \text{ a}^{-1}$  ( $\pm 96.2$  95% CI,  $R^2 = 0.043$ ,  $p > 0.05$ ), where the low  $R^2$  value indicates poor fit of the linear regression model to the Leone data series. There was a significant linear temporal trend in positions of the Nu'uuli and Masefau seaward margins. There was a highly variable nonlinear temporal trend in the position of the Leone seaward margin.

The Masefau mangrove seaward margin migrated landward about 3.0 m over the observed 42.1-year period, an average rate of  $63.9 \text{ mm a}^{-1}$ . The Leone mangrove seaward margin migrated landward about 9.3 m over the observed 42.7-year period, an average rate of  $24.5 \text{ mm a}^{-1}$ . The Nu'uuli mangrove seaward margin migrated landward about 138.1 m over the observed 42.2-year period, an average rate of  $72.3 \text{ mm a}^{-1}$ .

Fig. 4.16 shows the cumulative change in relative sea-level for the three mangrove sites based on mean annual sea-levels of the three years around the year of each remotely sensed image, adjusted for trends in sediment surface elevation for Nu'uuli mangrove. These results are used in the assessment of sea-level rise control on mangrove position, in Section 4.6.

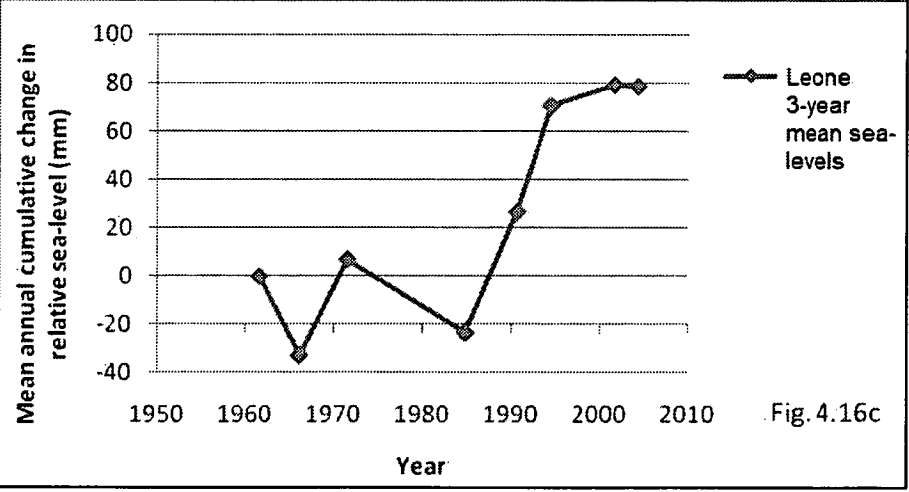
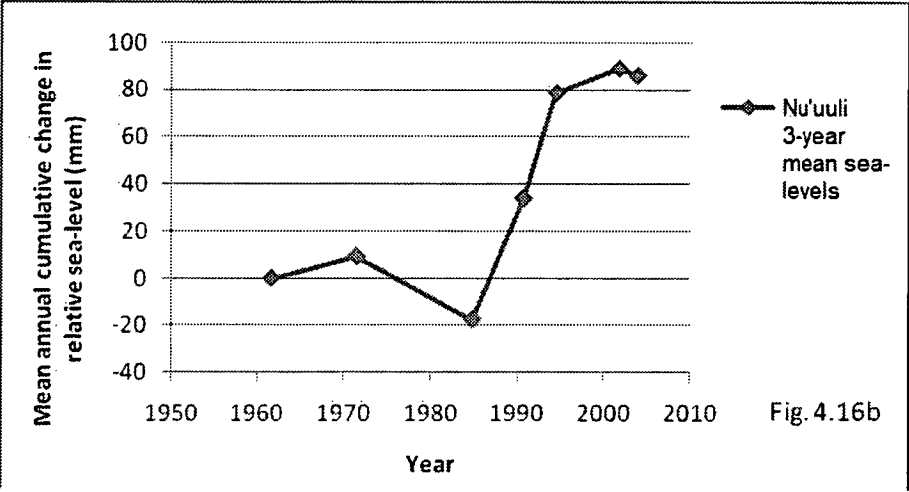
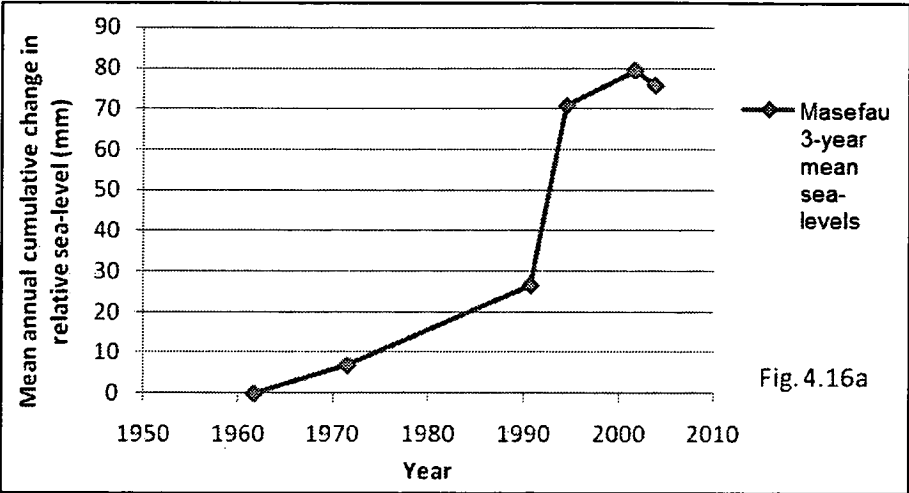


Fig. 4.16. Plots of cumulative change in relative sea-level versus date for (a) Masefau, (b) Nu'uuli and (c) Leone mangroves, American Samoa, based on mean annual sea-levels of the three years around the year of each date,



adjusted for trends in sediment surface elevation for Maseafu and Nu'uuli mangroves.

#### **4.4. TRENDS IN MANGROVE SURFACE ELEVATION**

When one researcher measured the height of a stake above the mangrove sediment surface ten times, this resulted in a standard deviation of the mean of  $\pm 1.2$  mm. When a second researcher conducted the same ten measurements, this resulted in a standard deviation of the mean of  $\pm 0.7$  mm. The difference between the two mean measurements was 1.1 mm and the standard deviation of the difference of means is 1.4, all within the same measurement precision.

Fig. 4.17 shows the temporal results of a VCGAMM fitted to the 1,412 sediment elevation data for the two study sites. This model comprised change in elevation as a function of site-specific temporal trend and spatial variation at the 33 sampling stations, while accounting explicitly both for stake- and station-specific heterogeneity. First, it is important to note the substantial uncertainty apparent in the estimated temporal effects. Fig. 4.17a shows the time-specific (temporal) trend in sediment elevation at Nu'uuli mangrove, where a linear declining trend over the sampling period was apparent, and no cyclical or seasonal trend in sediment surface elevation was apparent. Fig. 4.17b shows a significant nonlinear temporal trend in sediment elevation at the Masefau site over roughly the same sampling period, suggestive of a cyclical trend over 1.3 years. A VCGAMM that includes either an explicit effect accounting for the two study sites or an explicit factor for sampling station resulted in the same site-specific temporal trends, graphically summarized in Fig. 4.17.

Fig. 4.18 shows the spatial results of a VCGAMM fitted to the 1,412 sediment elevation data for the two study sites. Fig. 4.18a is based on using an explicit factor for sampling station, where each station is referred to according to the codes as shown in Figs. 3.1 and 3.2, with an M or N added before the station code to reflect if the station is located in Masefau (M) or Nu'uuli (N) mangrove study site. Similar to the temporal patterns presented in Fig. 4.17, Fig. 4.18a shows substantial uncertainty in the estimated spatial patterns for sampling station-specific trends. There was no significant

difference between the individual stations at each individual mangrove site, and also no significant difference between the 33 stations across the two sites. Fig. 4.18b, which used an explicit factor for study site, shows the site-specific effect in change in elevation, where there was a highly significant difference in the mean sediment elevation absolute change between the two sites — sediment elevation absolute change was significantly greater at Nu'uuli compared to Masefau (formal test of site-specific effect:  $N = 1412$ ,  $t$ -test statistic =  $-2.71$ ,  $P < 0.007$ ; Nu'uuli change in sediment surface elevation for site effect from the VCGAMM shown in Fig. 4.18b estimate of  $-3.432 \text{ mm a}^{-1}$ ,  $\pm 1.267 \text{ SE}$ ). The Nu'uuli point estimate and error interval are relative to the Masefau reference level, so the Masefau 0 reference level is a baseline with no error associated with it (Wood, 2006).

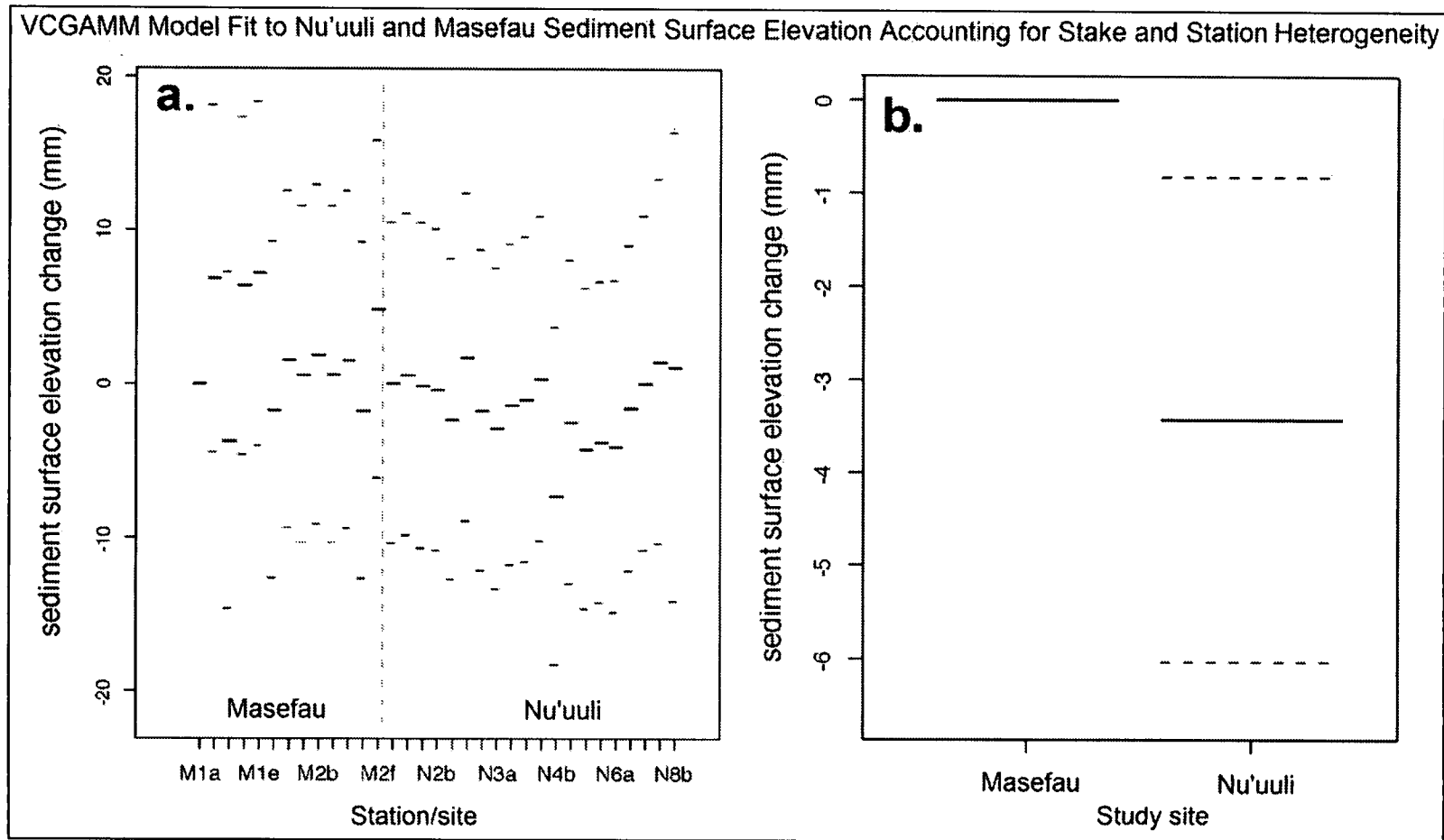


Fig. 4.18 Graphical summary of varying coefficient generalized additive mixed regression model (VCGAMM) fitted to sediment surface elevation change measurements for Nu'uuli and Masefau mangroves, American Samoa, to show spatial patterns. Sampling stations and elevation stakes within each of the two sampling sites are modeled as random effects while study site is modeled as a fixed effect. (a) shows sampling station-specific spatial trend in change in measured surface sediment elevation at the two study sites, where there is an explicit effect accounting for sampling station. (b) shows site-specific effect in change in elevation, where there is an explicit effect accounting for the two study sites. Solid bars = mean, dashed bars = 95% confidence interval, y-axis = centered response scale. In (a), the first sampling station (M1a) is the reference level and so is centered at zero. In (b) the Masefau site is the reference level and so is centered at zero.

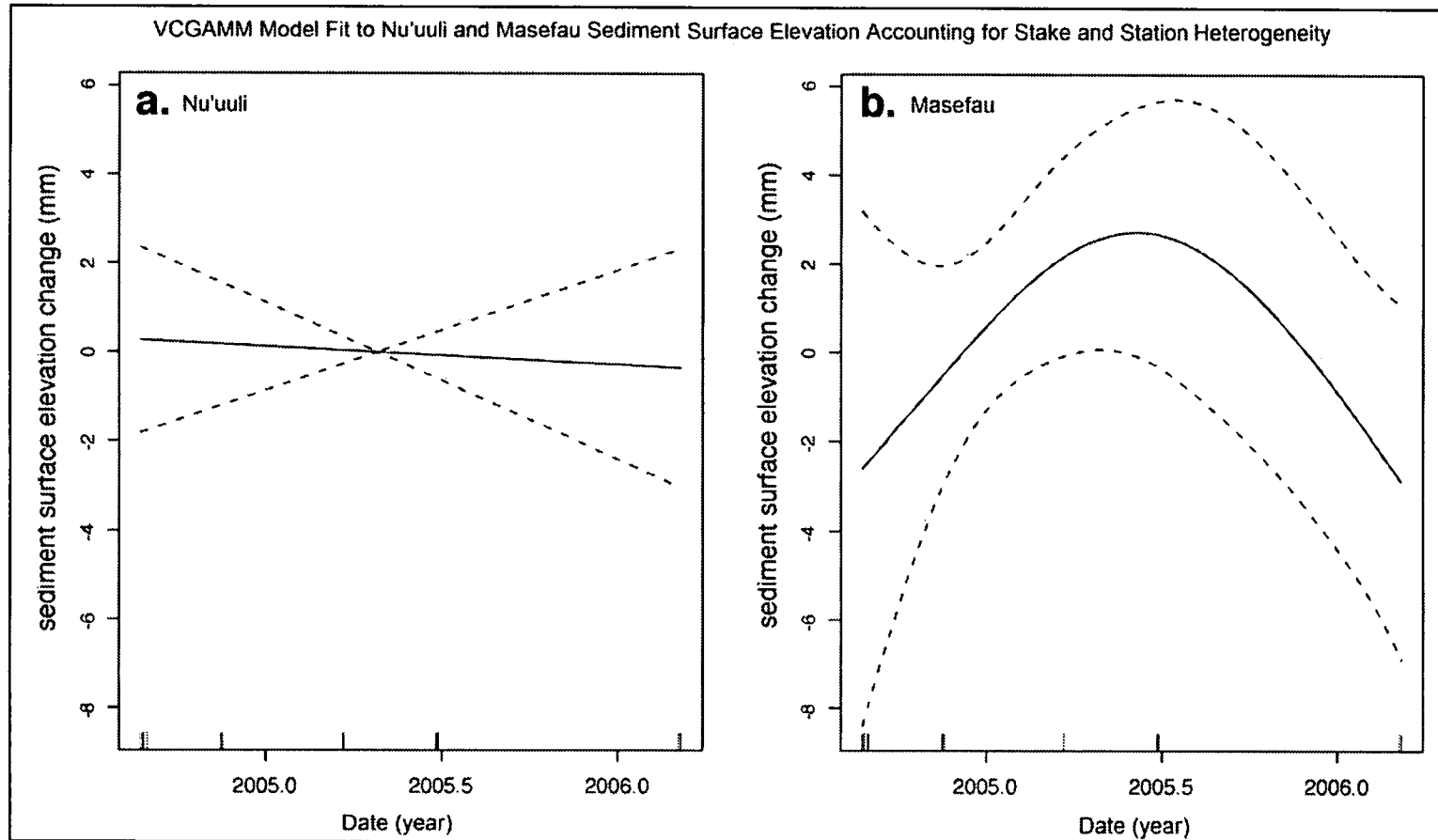


Fig. 4.17. Graphical summary of varying coefficient generalized additive mixed regression model (VCGAMM) fitted to sediment surface elevation change measurements for Nu'uuli and Masefau mangroves, American Samoa, to show temporal patterns. Sampling stations and elevation stakes within each of the two sampling sites are modeled as random effects while study site is modeled as a fixed effect. (a) shows a linear site-specific temporal trend in change in elevation at the Nu'uuli site across all sampling stakes and stations from 25 August 2004 to 6 March 2006. (b) shows a significant nonlinear site-specific temporal trend in change in measured elevation at Masefau across all sampling stakes and stations from 17 November 2004 to 8 March 2006. Solid curves = model fit, dashed curves = 95% pointwise confidence bands, y-axis = centered response scale. The vertical bars on the topside of the lower x-axis of panels are known as a 'rug', which shows the data distribution within each panel.

The time-specific linear trend estimate for Nu'uuli from the VCGAMM accounting explicitly for both stake- and station-specific heterogeneity was  $-0.246 \text{ mm a}^{-1}$  ( $\pm 0.95 \text{ SE}$ ), where a negative sign indicates lowering in the sediment surface elevation, recognizing that there is very high uncertainty in this estimate (Fig. 4.17a). There was a ca.  $-0.577 \text{ mm a}^{-1}$  ( $\pm 1.04 \text{ SE}$ ) decline in sediment surface elevation using a standard linear regression model that does not account for the sampling design heterogeneity; the confidence intervals resulting from the two techniques overlap.

A linear temporal trend estimate is not valid for Masefau, as the trend was a significant nonlinear function suggestive of a cyclical effect ( $P < 0.05$ ) with no net change over the 1.3 year sampling period (Fig. 4.17b). Based on this result,  $0 \text{ mm a}^{-1}$  (no linear trend) is the best estimate of the net change in Masefau's sediment surface elevation over periods of past and projected future decades, recognizing that there is very high uncertainty in this estimate (Fig. 4.17b) and that a longer period of observation, potentially over decades, could reduce the uncertainty.

#### **4.5. MANGROVE SEDIMENT ACCRETION RATES FROM $^{137}\text{Cs}$ AND EXCESS $^{210}\text{Pb}$ ACTIVITY DATING**

The core from Nu'uuli recovered a 45 cm profile with compression of 0.5 cm (1% compression). The core at Masefau recovered a 46.4 cm profile with compression of 2.3 cm (5% compression). Both cores were organic silty clay, similar to those described from these sites by Ellison (2001).

Fig. 4.19 shows average  $^{137}\text{Cs}$  activity in 1 cm core sections by depth for Nu'uuli mangrove. Concentrations of  $^{137}\text{Cs}$  are low and generally variable. A  $^{137}\text{Cs}$  concentration peak is discernible at 15.5 cm depth (the average  $^{137}\text{Cs}$  concentration in the core at 15-16 cm depth), assumed to correspond to the year of peak  $^{137}\text{Cs}$  deposition in 1963. This indicates the mangrove has experienced a  $3.8 \text{ mm a}^{-1}$  average sediment accretion rate over the 41-year period between peak  $^{137}\text{Cs}$  deposition and date of core extraction. The  $^{137}\text{Cs}$  activities at depth  $\geq 25 \text{ cm}$  are considerable and more prominent than in the region 0-17 cm. For the purpose of subsequent analysis in this thesis, the suggested accretion rate of  $3.8 \text{ mm a}^{-1}$  is

accepted, which if valid, would mean that the  $^{137}\text{Cs}$  presence at lower depth was a result of disturbance before extraction. (Section 5.5.2 explains why there is low confidence in the resulting accretion rate estimates). For Nu'uuli mangrove, the rate of change in sediment elevation due to surface processes is based on the observed depth of peak  $^{137}\text{Cs}$  activity at 15.5 cm depth, accounting for about the upper 10% of the mangrove sediment profile.

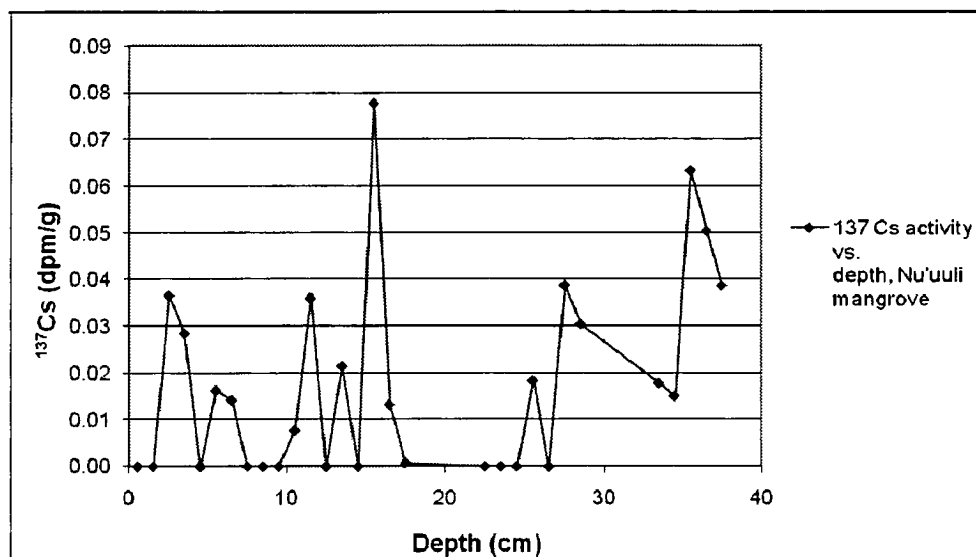


Fig. 4.19.  $^{137}\text{Cs}$  activity depth profile for Nu'uuli mangrove.

Fig. 4.20 shows a semi-log excess  $^{210}\text{Pb}$  decay curve from 0-27 cm depth for Maséfau mangrove, excluding data assumed to be corrupted by bioturbation or other disturbance, indicating an average sediment accretion rate of  $3.3 \pm 0.3 \text{ mm a}^{-1}$  ( $R^2 = 0.80$ ,  $n = 19$ ) over the past 82 years. For Maséfau mangrove, the rate of change in elevation due to surface processes is based on the observed lower depth of excess  $^{210}\text{Pb}$  activity at 27 cm depth, accounting for about the upper 20% of the mangrove sediment profile.

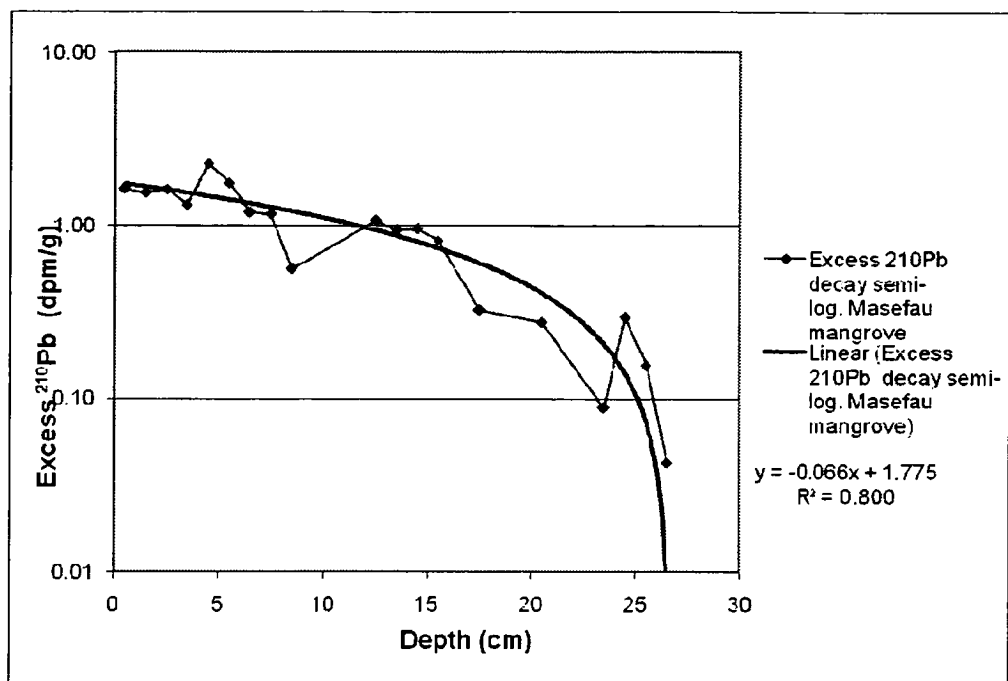


Fig. 4.20. Excess  $^{210}\text{Pb}$  decay semi-log plot for Masefau mangrove from 0-27 cm depth with linear regression model fit to the data.

Due to disturbance to the sediment profiles, the Nu'uuli core analysis did not produce a coherent excess  $^{210}\text{Pb}$  depth profile and the Masefau core did not produce a coherent  $^{137}\text{Cs}$  activity depth profile.

#### 4.6. RELATIVE SEA-LEVEL CONTROL ON MANGROVE POSITION

Using the full 55.6-year Pago Pago tide gauge data series, for Masefau mangrove, there is a highly significant positive correlation between mangrove area and change in relative sea-level over the observed 42.1-year period of remotely sensed imagery, based on linear regression analysis ( $P < 0.01$ ,  $R^2 = 0.913$ ,  $N = 6$ ). Nu'uuli mangrove also demonstrated a highly significant correlation over the observed 42.2-year period ( $P < 0.01$ ,  $R^2 = 0.957$ ,  $N = 7$ ). For Leone mangrove, the correlation over the observed 42.7-year period is not significant ( $P > 0.05$  [ $P = 0.6$ ],  $R^2 = 0.043$ ,  $N = 8$ ).

Using the rate of change in site-specific relative sea-level for Nu'uuli mangrove (Section 4.7), again employing the full tide gauge 55.6 year period, there again is a highly significant correlation over the observed 42.1-year

period ( $P < 0.01$ ,  $R^2 = 0.955$ ,  $N = 7$ ). Because there was no linear trend in sediment surface elevation for Masefau mangrove (Section 4.4), Masefau's site specific relative sea-level rise rate is equivalent to the Pago Pago tide gauge regional relative sea-level rise rate.

Fig. 4.21 is a plot of change in mangrove area versus change in relative sea-level using the average of mean annual sea-levels of the three years around the year of each remotely sensed image, adjusted for trends in sediment surface elevation for Nu'uuli mangrove, with a linear regression model fit to the data series. Nu'uuli mangrove demonstrated a significant correlation over the observed 42.2-year period ( $P < 0.05$ ,  $R^2 = 0.60$ ,  $N = 7$ ). For Masefau mangrove, there is a highly significant correlation between change in mangrove area and change in mean annual relative sea-level over the observed 42.1-year period of remotely sensed imagery, based on linear regression analysis ( $P < 0.01$ ,  $R^2 = 0.89$ ,  $N = 6$ ). For Leone mangrove, once again, the correlation over the observed 42.7-year period is not significant ( $P > 0.05$  [ $P = 0.7$ ],  $R^2 = 0.020$ ,  $N = 8$ ).



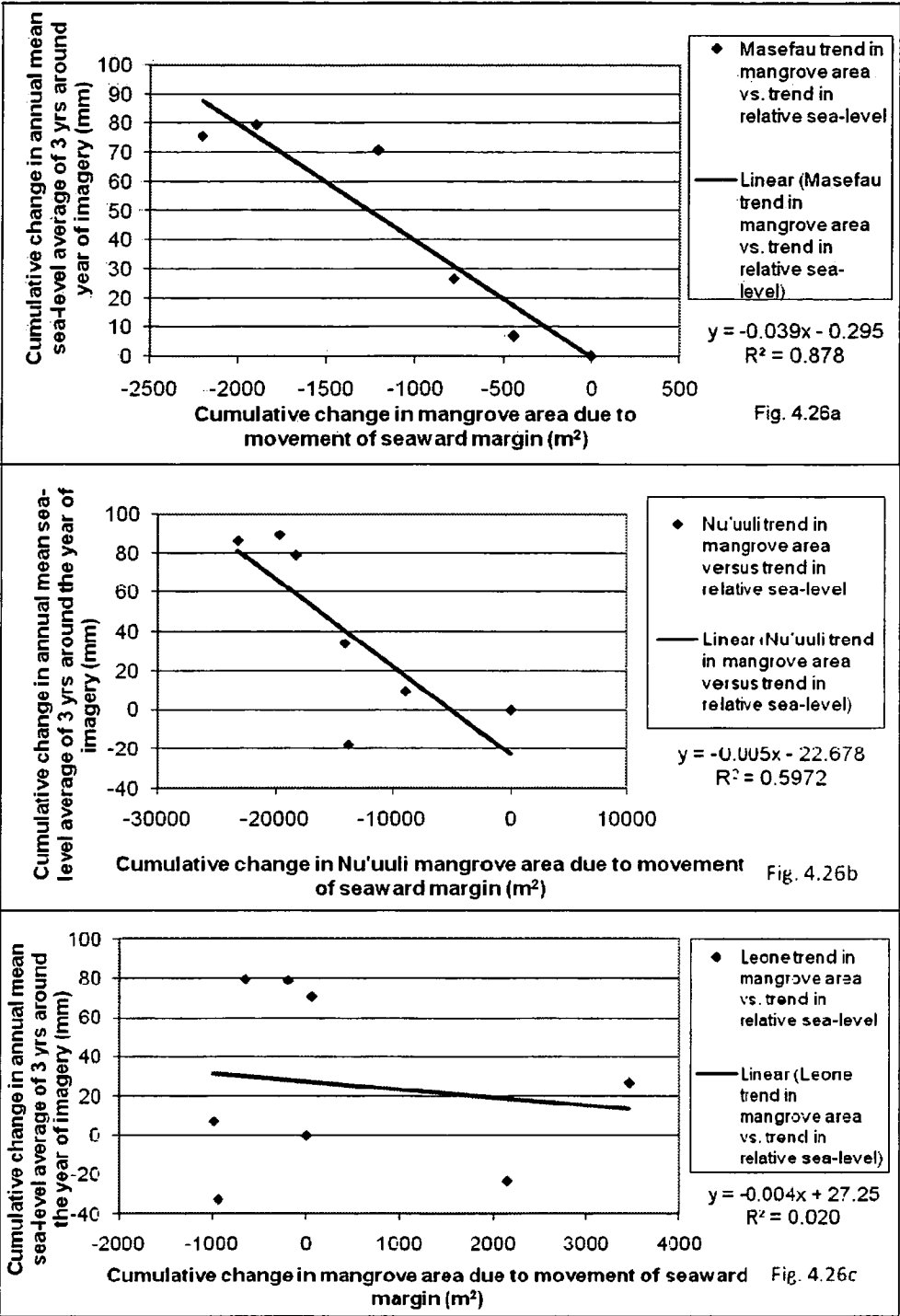


Fig. 4.21. Plot of the change in mangrove area resulting from movement of the seaward mangrove margin versus trend in relative sea-level using mean annual sea-levels from the Pago Pago, American Samoa tide gauge, and linear regression model for (a) Masefau, (b) Nu'uuli and (c) Leone

mangroves, American Samoa from 1961-2003/4. Sea-levels are adjusted for trends in sediment surface elevation for Nu'uuli mangrove.

**4.7. MANGROVE RESISTANCE TO CHANGES IN RELATIVE SEA-LEVEL**

Nu'uuli mangrove has experienced site-specific relative sea-level rise of 2.22 ( $\pm$  2.22 95% CI) mm a<sup>-1</sup>, based on an observed regional relative sea-level rise rate of 1.97 ( $\pm$  0.32 95% CI) mm a<sup>-1</sup>, and a rate of change in sediment elevation of -0.25 ( $\pm$  1.9 95% CI) mm a<sup>-1</sup>. Fig. 4.22 shows that the IPCC projections applied to Nu'uuli mangrove range between 171 and 938 mm rise in relative sea-level from 1990 to 2100, and a rise of between 152 and 885 mm between 2004 and 2100.

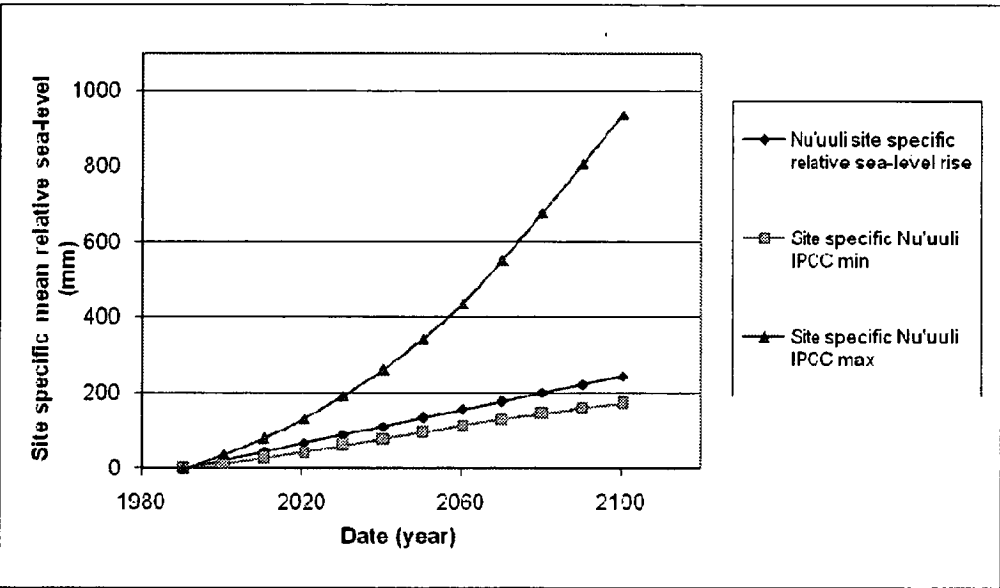


Fig. 4.22. Linear regression model fit to site specific monthly mean relative sea-levels for Nu'uuli mangrove, 1948-2004, and plots of upper and lower projections for the change in site specific relative sea-level adjusted from IPCC global projections from 1990 through 2100.

Masefau mangrove has experienced site-specific relative sea-level rise equivalent to the observed regional relative sea-level rise rate of 1.97  $\pm$  0.32 mm a<sup>-1</sup>, based on there being no observed linear trend in change in sediment elevation (0 mm a<sup>-1</sup>). Fig. 4.2, which shows the IPCC projections

for American Samoa range between a rise of between 64 mm and 831 mm between 2004 and 2100 (Church et al., 2001), therefore applies to Masefau mangrove, where no correction for site specific relative sea-level change is needed.

Nu'uuli mangrove has likely been experiencing site-specific relative sea-level rise based on the difference between the rate of change in regional relative sea-level and rate of change in sediment surface elevation, where the regional relative sea level rise rate of  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI) ( $1.65$  to  $2.29 \text{ mm a}^{-1}$ ) lower interval just touches the upper error interval of the rate of change in elevation of the mangrove surface of  $-0.25 \text{ mm a}^{-1}$  ( $\pm 1.90$  95% CI) ( $-2.15$  to  $1.65 \text{ mm a}^{-1}$ ).

Masefau mangrove has been experiencing site-specific relative sea-level rise of  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI). This is based on there being a significant difference between the rate of change in regional relative sea-level and observation of no linear trend in sediment surface elevation: Based on non-overlapping error intervals, the regional relative sea level rise rate of  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI) has been exceeding the  $0 \text{ mm a}^{-1}$  lack of change in elevation of the mangrove surface, which would have caused the mangrove to retreat landward over past decades.

Masefau is already not keeping pace. Nu'uuli may already not be keeping pace, and any increase in its site specific rate of sea-level rise would cause there to be a significant elevation deficit. Both Nu'uuli and Masefau mangroves will not keep pace with the IPCC upper projection applied to American Samoa of 831 mm over 96 years, equivalent to an average linear rate of rise of  $8.7 \text{ mm a}^{-1}$ . This projected rise would include an acceleration term, thus this reported rate of average linear rise is presented for the purpose of comparison with the sediment elevation rates. This analysis was not performed for Leone mangrove, where monitoring trends in elevation of the mangrove sediment surface was not conducted.

#### **4.8. RESILIENCE TO RELATIVE SEA-LEVEL RISE: PREDICTED CHANGE IN MANGROVE POSITION**

Table 4.2 presents predicted reductions in area of the three mangrove study sites as a result of projected landward migration of the seaward mangrove margin through the middle of the year 2100, and reports the rate of change in area. For Masefau and Nu'uuli, sites shown to have highly significant correlations between mangrove area and observed change in relative sea-level, three scenarios for change in site-specific relative sea-level are assessed to predict year 2100 change in mangrove area resulting from movement of the seaward mangrove margin:

- (i) Lower projection for change in site specific relative sea-level adjusted from the IPCC lower global projection from 1990 through 2100;
- (ii) Upper projection for change in site specific relative sea-level adjusted from the IPCC upper global projection;
- (iii) Extrapolation from the observed recent historical erosion rate of the mangrove seaward margin, which produces the same result as extrapolating through the year 2100 from the linear regression model fit to the observed trend in site specific relative sea-level (based on measurements of the rate of change in elevation of the mangrove sediment surface and rate of change in regional relative sea-level from observed mean monthly relative sea-levels from the Pago Pago tide gauge).

For Leone, where there was no significant correlation between change in mangrove area resulting from migration of the seaward mangrove margin and change in relative sea-level, only the method of extrapolating through the year 2100 from the observed recent historical erosion rate of the mangrove seaward margin is employed.

Table 4.2. Scenarios for change in mangrove area resulting from migration of the seaward mangrove margin, and rate of movement of the seaward mangrove margin, through the middle of 2100.

Mangrove site	Reduced mangrove area by mid-2100 (m <sup>2</sup> ) <sup>a</sup>	Annual rate of reduction in mangrove area (m <sup>2</sup> a <sup>-1</sup> )	Rate of landward migration of seaward mangrove margin (mm a <sup>-1</sup> )
Leone <sup>b</sup>			
Extrapolate observed historical erosion rate	1,949	20.3	24.5
Masefau <sup>c</sup>			
Adjusted IPCC low projection	1,700	17.7	21.6
Extrapolate observed historical erosion rate/ observed trend in site specific relative sea-level	5,024	52.3	63.9
Adjusted IPCC upper projection	22,077	230.0	280.8
Nu'uuli <sup>c</sup>			
Adjusted IPCC low projection	37,577	391.4	51.6
Extrapolate observed historical erosion rate/observed trend in site specific relative sea-level	52,686	548.8	72.3
Adjusted IPCC upper projection	218,786	2,279.0	314.8

<sup>a</sup> Predicted change in mangrove area from mid-2004 through mid-2100, an elapsed period of 96 years.

<sup>b</sup> Extrapolation from observed erosion rate. The correlation between change in mangrove area from movement of the seaward margin and change in regional relative sea-level was not significant for this study site.

<sup>c</sup> Extrapolation employing three scenarios for projected site specific relative sea-level rise. There was a highly significant correlation between change in mangrove area due to movement of the seaward margin and change in relative sea-level for these two study sites.

The slope of the upland adjacent to the landward mangrove margin at Leone mangrove is a mean of 6% (standard deviation of the mean = 2%, N = 25). Approximately 23.4% of the Leone landward mangrove margin is obstructed from natural landward transgression. The slope of the upland adjacent to the landward mangrove margin at Masefau mangrove is a mean of 27% (standard deviation of the mean = 6%, N = 25). Approximately 16.5% of the Masefau landward mangrove margin is obstructed from natural landward transgression. The slope of the upland adjacent to the landward mangrove margin at Nu'uuli mangrove is a mean of 7.7% (standard deviation of the mean = 2%, N = 128). Approximately 68% of the Nu'uuli landward mangrove margin is obstructed from natural landward transgression.

Table 4.3 presents predicted increases in mangrove area through the year 2100 resulting from the landward migration of the landward margins of the three study sites.

Table 4.3. Scenarios for change in mangrove area resulting from migration of the landward mangrove margin, and rate of movement of the landward mangrove margin, through the middle of 2100.

Mangrove site	Increased mangrove area by mid-2100 (m <sup>2</sup> ) <sup>a</sup>	Annual rate of increase in mangrove area (m <sup>2</sup> a <sup>-1</sup> )	Rate of landward migration of landward mangrove margin (mm a <sup>-1</sup> )
Leone			
IPCC low projection			
Extrapolate observed trend in regional relative sea-level from American Samoa tide gauge	1,040	10.8	11.1
IPCC upper projection	3,074	32.0	32.8
	13,505	140.7	144.3
Masefau			
Adjusted IPCC low projection			
Extrapolate from observed trend in site specific relative sea-level	252	2.6	2.5
	745	7.7	7.3
Adjusted IPCC upper projection	3,269	34.1	32.1
Nu'uuli			
Adjusted IPCC low projection			
Extrapolate from observed trend in site specific relative sea-level	4,057	42.3	20.6
	5,689	59.3	28.8
Adjusted IPCC upper projection	23,623	246.1	119.7

<sup>a</sup> Change in mangrove area from mid-2004 through mid-2100, an elapsed period of 96 years.

If the observed trend in regional and site specific relative sea-level over the previous 55 years continue through the year 2100, and no new obstacles to landward migration are constructed, based on these estimated movements of the landward and seaward margins, Leone mangrove will experience a net gain in area of 1,125 m<sup>2</sup>, while Masefau and Nu'uuli mangroves will experience a net loss in area of 4,279 m<sup>2</sup> and 46,997 m<sup>2</sup>, respectively. This translates to a net 11.7% reduction in combined mangrove area under this scenario for projected change in site specific and regional relative sea-level.

Applying the IPCC's upper projection for sea-level rise by the year 2100, if no new obstacles to natural mangrove migration are constructed, Leone mangrove will increase in area by 11,556 m<sup>2</sup>, while Masefau and Nu'uuli mangroves will lose 18,808 m<sup>2</sup> and 195,173 m<sup>2</sup>, respectively. This translates to a net 47.3% reduction in combined mangrove area under this scenario for projected change in site specific and regional relative sea-level. Where unobstructed by development, by the year 2100, the landward mangrove margins of Leone, Masefau, and Nu'uuli could migrate landward as much as 13.8 m, 4.0 m, and 12.0 m, respectively, under IPCC's upper projection.

#### **4.9. REGIONAL EXTRAPOLATION**

Table 4.4 provides an approximate, first-order prediction of the change in mangrove area through the year 2100, based on observations from American Samoa and several assumptions (Section 3.10), for countries and territories of the Pacific Islands region where mangroves are indigenous. Based on extrapolating relative sea-level trends through the year 2100, the current estimated 516,469 ha of indigenous mangroves in the Pacific Islands region would be reduced by 2.4%. Under IPCC's upper projection, Pacific Island mangrove area could be reduced by 22.4%. Under the first scenario, of no acceleration in change in sea-level, 7 of the 16 countries and territories would experience a loss in mangrove area. Under the upper projection, 14 of the countries and territories would experience losses, with only Samoa and the Solomon Islands experiencing no losses.

Table 4.4. Rough estimate of mangrove response to relative sea-level change for the 16 Pacific Island countries and territories where mangroves are indigenous.

Tide Gauge Station Name and Country or territory	Mangrove area (ha) most current estimate	Observed regional relative sea-level change rate (mm a <sup>-1</sup> ) <sup>a</sup>	Observed or estimated rate of change in elevation of mangrove sediment surface (mm a <sup>-1</sup> ) <sup>b</sup>	Rate of change in site specific relative sea-level through year 2100 (mm a <sup>-1</sup> )		Year 2100 mangrove area extrapolating historic relative sea-level trends (ha)	Year 2100 mangrove area applying IPCC upper projection (ha)
				Extrapolate from historical record	Applying IPCC upper projection		
Pago Pago, American Samoa	52 <sup>c</sup>	+1.97 <sup>a1</sup>	-0.125	+2.095	+8.925	45.92	27.40
Saipan, Northern Mariana Islands	5 <sup>d</sup>	+0.9 <sup>a4</sup>	+0.5	+0.4	+6.71	4.89	3.24
Pohnpei, Federated States of Micronesia	8,564 <sup>e</sup>	+1.8 <sup>a2</sup>	+1.3	+0.5	+6.81	8,324.86	5,504.42
Suva, Fiji	41,000 <sup>f</sup>	+6.7 <sup>a2</sup>	+1.5	+5.2	+11.51	29,093.37	16,241.73
Guam	70 <sup>g</sup>	-0.6 <sup>a2</sup>	+0.5	-1.1	+5.21	70	50.87
Kanton Island, Kiribati	258 <sup>h</sup>	-0.4 <sup>a2</sup>	+0.5	-0.9	+5.41	258	184.78
Majuro, Marshall Islands	4 <sup>i</sup>	+2.8 <sup>a2</sup>	+0.5	+2.3	+8.61	3.49	2.19
Nauru	1 <sup>g</sup>	-1.94 <sup>a3</sup>	+0.5	-2.44	+3.87	1	0.80
Noumea, New Caledonia	20,250 <sup>j</sup>	+0.2 <sup>a2</sup>	+1.5	-1.3	+5.01	20,250	14,927.95
Malakal, Palau	4,500 <sup>k</sup> 372,770 <sup>g</sup>	+1.0 <sup>a2</sup> -0.73 <sup>a3</sup>	+1.5 +1.5	-0.5 -2.23	+5.81 +4.08	4,500 372,770	3,128.44 292,988.95



Port Moresby, Papua  
New Guinea

Apia, Samoa	700 <sup>i</sup>	-5.0 <sup>a4</sup>	+1.5	-6.5	-0.19	700	700
Honiara, Solomon Islands	64,200 <sup>m</sup>	-7.0 <sup>a2</sup>	+1.5	-8.5	-2.19	64,200	64,200
Nuku'alofa, Tonga	1,305 <sup>n</sup>	+1.3 <sup>a4</sup>	+0.5	+0.8	+7.11	1,246.70	818.23
Funafuti, Tuvalu	40 <sup>g</sup>	+2.3 <sup>a2</sup>	+0.5	+1.8	+8.11	35.98	22.98
Port Vila, Vanuatu	2,750 <sup>g</sup>	+1.0 <sup>a4</sup>	+1.5	-0.5	+5.81	2,750	1,911.82

<sup>a1</sup> Refer to Section 4.1.

<sup>a2</sup> Calculated from fitting a linear regression model to monthly mean relative sea-levels from historical tide gauge records with tidal constituents removed.

<sup>a3</sup> Calculated from fitting a linear regression model to monthly mean relative sea-levels. Port Moresby, Papua New Guinea has < 20-year tide gauge record but a relative sea-level change rate from reconstructed analysis was not available.

<sup>a4</sup> For these locations, with a local tide gauge record of < 20 years, relative sea-level change trends are calculated from TOPEX/Poseidon satellite altimetry data combined with historical global tide gauge records over the period 1950-2001 employing the method by Church et al. (2004a).

<sup>b</sup> American Samoa from this study; for Federated States of Micronesia, average of observed rates from plots in *Rhizophora* spp., *Sonneratia alba*, and *Bruguiera gymnorhiza* mangrove stands in the Enipoas River basin, Pohnpei, measured using stakes driven to a depth of 70 cm observed over 2.5 years (Krauss et al., 2003). Does not account for subsurface processes occurring below 70 cm. For areas lacking data on mangrove sediment elevation rates, we make the rough estimate that low island mangroves have a +0.5 mm a<sup>-1</sup> rate of rise in sediment elevation, and high island mangroves the rate is estimated to be +1.5 mm a<sup>-1</sup> based on available information from Cahoon and Hensel (2006); Krauss et al. (2003) and results from this study for American Samoa.

<sup>c</sup> Bardi and Mann (2004).

<sup>d</sup> Gilman (1998, 1999b).

<sup>e</sup> MacLean et al. (1998), compilation of previous assessments interpreting aerial photography from 1976 combined with 1983 fieldwork.

<sup>f</sup> Watling (1985).

<sup>g</sup> Scott (1993). For Papua New Guinea, estimate is between 353,770 – 391,770 ha. For Vanuatu, estimate is between 2,500 – 3,000 ha.

<sup>h</sup> Nenenteiti Teariki-Ruatu, personal communication, February 2005, Republic of Kiribati Ministry of Environment, Lands, and Agricultural Development.

<sup>i</sup> John Bungitak, personal communication, August 2005, Environmental Protection Agency, Republic of the Marshall Islands.

<sup>j</sup> Thollot (1987).

<sup>k</sup> Maragos (1994); Metz (2000).

<sup>l</sup> Pearsall and Whistler (1991).

<sup>m</sup> Hansell and Wall (1976).

<sup>n</sup> Wiser et al. (1999).

## **Chapter 5**

### **Discussion**

#### **American Samoa Mangrove Vulnerability and Responses to Relative Sea-Level Rise**

##### **5.1. TRENDS IN AMERICAN SAMOA RELATIVE SEA-LEVEL**

Determining the linear trend in relative sea-level measured by a tide gauge proximate to a mangrove study site, or otherwise from reconstructed analysis using satellite altimetry data, provides one component of information needed to determine how sea-level has been changing relative to the mangrove sediment surface, and ultimately, to assess vulnerability to any sea-level rise (Sections 1.5.6, 1.5.7). In the plot of mean monthly relative sea-levels from the Pago Pago, American Samoa tide gauge, the influence of the El Nino phase of the El Nino-Southern Oscillation is evident as large negative departures from the run of mean sea-level values, amongst a long-term rising trend in relative sea-level (Fig. 4.1). Large negative spikes in relative mean sea-level series are known to be associated with ocean circulation changes during El Nino events in the Pacific (Hunter, 2002; Woodworth and Blackman, 2004; Church et al., 2006).

Mean monthly relative sea-level data, calculated by averaging hourly sea-levels by month, adequately removes cyclical tidal constituents (personal communication, Dr. Philip Woodworth, Proudman Oceanographic Laboratory, June, 2005). The Pago Pago relative sea-level trend has large interannual variability due to El Nino Southern Oscillation (ENSO) events (Church et al., 2006). Removal of ENSO phase signals through a filtering method (e.g., spectral filtering of the tide gauge data) may have resulted in the data fitting better to the regression model, a better estimate for the trend in change in relative sea-level and a smaller confidence interval around the point estimate of the trend in relative sea level rise. For instance, Church et al. (2006) calculated relative sea-level rise from the Pago Pago tide gauge data series to be  $1.6 - 2.1 \text{ mm a}^{-1}$ , after removing the SOI/NINO3 index and conducting

other corrections to address serial correlation, which includes the point estimate derived from employing the simple linear regression model. The assumption of independence of monthly mean sea level values employed in the regression model used here is likely false, as there is serial correlation in the monthly tide gauge data. A better estimate of the effective number of degrees of freedom in the time series could be made to address this serial correlation.

The individual components contributing to the observed rate of relative sea-level rise measured by the Pago Pago tide gauge are absolute sea-level rise, glacial isostatic adjustment (GIA, Section 1.4), atmosphere pressure effects (inverse barometer, IB), and land motion (Woppelmann et al., 2007), where we can write in simple terms:

$$\text{absolute sea-level} = \text{relative sea-level} - \text{GIA\&IB} + \text{land motion}$$

In this equation, the 'land motion' term account for all factors affecting the island's vertical position (e.g., tectonic uplift or subsidence, coastal subsidence) but excludes GIA, included as a separate term in the equation. The rate of change of Pacific Ocean mean (absolute) sea-level, based on 21 tide gauge records with  $\geq 24$  year length records in the 1950-2001 period, is estimated to be  $1.5 \text{ mm a}^{-1}$ , after making GIA/IB corrections, consistent with the global average rate (Section 1.4) (Church et al., 2006; Bindoff et al., 2007; Solomon et al., 2007).

GIA/IB effects are estimated to contribute  $-0.5 \text{ mm a}^{-1}$  in the Pacific Islands region (Church et al., 2006). The GIA/IB value is from Church et al. (2006) (Table 1 entry for the Pago Pago, American Samoa tide gauge), where their rate of relative sea-level rise of  $1.6 \text{ mm a}^{-1}$  (for the period January 1950 – August 2000) is subtracted from the tide gauge trend corrected for GIA/IB of  $2.1 \text{ mm a}^{-1}$ . Different GIA models can result in disparate predictions at the required level of a few tenths of millimeters per year accuracy, and other processes that may affect the vertical stability of tide gauges are more difficult to predict (Woppelmann et al., 2008).

Given the uncertainties in the relative sea-level trends of  $\pm 0.2\text{-}1.2 \text{ mm a}^{-1}$  from Church et al.'s (2006) Pacific region analysis, and the large

uncertainties associated with effects such as post-emergence contamination of dated paleoshoreline features and a limited continuous GPS time series near the Pago Pago tide gauge station, no accurate estimate can be made about land motion at the Pago Pago tide gauge site. As a result, measuring rather than modeling vertical land motion at tide gauges is necessary in order to determine accurate estimates of absolute sea-level. Estimates of land motion of Tutuila Island, American Samoa are possible from dated emergent paleoshoreline features, reviewed in Section 1.4.4. Possible concerns with post-emergence contamination of the Tutuila paleoshoreline features have been raised (Nunn, 1998a). Continuous GPS stations located adjacent to tide gauges can also provide estimates of land motion, however, at least 10 years of high-quality time series data are needed before any reliable land motion estimates can be derived. Currently, the length of the continuous GPS data time series in American Samoa is inadequate to make definitive conclusions. Based on less than five years of GPS data collected to date, relatively high land subsidence is apparent for American Samoa (the GPS is located 6.3 km from the Pago Pago tide gauge station) ( $-5.24 \pm 0.33 \text{ mm a}^{-1}$ ) ([http://www-gpsg.mit.edu/~tah/MIT\\_IGS\\_AAC/index2.html](http://www-gpsg.mit.edu/~tah/MIT_IGS_AAC/index2.html), station ASPA [American Samoa Power Authority] accessed 25 March 2008).

## **5.2. TRENDS IN FREQUENCY AND ELEVATION OF EXTREME HIGH WATER EVENTS**

Extreme high water event levels are projected to increase over coming decades as a result of the same atmospheric and oceanic factors that are causing global sea-level to rise, and possibly also as a result of variations in regional climate (Section 1.3.1) (Hunter, 2002; Woodworth and Blackman, 2004; Church et al., 2001, 2004b). Of the range of climate change factors (Section 1.3), changes in extreme high waters are directly linked to changes in relative sea-level.

In American Samoa, changes in extreme high waters are likely to have a small effect on mangrove area and health relative to the effect of relative mean sea-level rise and non-climate related anthropogenic stressors. Results indicate that the frequency and return period of extreme high water events have had no significant linear temporal trend. Extreme water levels

have been increasing at about  $1.9 \text{ mm a}^{-1}$ , within the error intervals of the rates of observed increase in relative mean and median sea-levels, indicating that this climate change outcome is not likely to pose an increasing threat to American Samoa mangroves beyond the effects from rising sea-level.

It is not well understood how changes in extreme high waters affect mangroves. Increased levels and frequency may result in similar effects as observed to be caused by storm events, including tree mortality and stress, sulphide soil toxicity, and altered sediment elevation (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Smith et al., 1994; Mastaller, 1996; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Baldwin et al., 2001; Sherman et al., 2001; Woodroffe, 1995b, 2002; Cahoon et al., 2003, 2006; Cahoon, 2006; Piou et al., 2006).

#### **5.2.1. Factors Causing Increase in Extreme High Water Event Levels**

Point estimates for trends in change of monthly means, annual means, annual medians, highest 0.1% hourly, and highest 0.05% hourly relative sea-levels fall within a narrow  $0.26 \text{ mm a}^{-1}$  range, and 95% CI of the slopes from linear regression models all overlap. Because extreme high water levels are increasing at rates that are not significantly different from that of increases in relative mean and median sea-levels, this indicates that similar factors that are producing change in median and mean sea-level are also causing change in extreme high water levels to the same degree. Serial correlation in the tide gauge data, such as from seasonal cycles, likely have produced artificially small 95% CIs. There has been no removal of cyclical tidal signals or phases of the El Nino Southern Oscillation beyond the removal of some background noise through averaging. Statistical analysis with a better estimate of the effective number of degrees of freedom in the time series would produce larger confidence intervals around the point estimates. However, the confidence intervals would still all overlap with this improved analysis.

The American Samoa mean monthly relative sea-level series (Fig. 4.1) and to a certain degree the higher percentile series (Figs. 4.9 – 4.10) and plots of just the annual 99.9 and 99.95 percentiles (not shown) show large negative spikes, known to be associated with El Nino events (Hunter, 2002;

Woodworth and Blackman, 2004). Subtracting the median (the 50-percentile) removed this common signal, largely confirming that in American Samoa extreme high water events are not linked to El Nino beyond the degree to which this factor influences change in median and mean sea-level. In other words, while the variations in extreme high water levels are apparently correlated with changes in regional climate, the variability in these extremes are similar to those in mean and median sea-level, indicating extreme high water events and mean sea-level have the same magnitude and type of atmospheric and oceanic forcing.

The results from American Samoa are generally consistent with those of D'Onofrio et al. (1999), Woodworth and Blackman (2002, 2004) and Hunter (2002). D'Onofrio et al. (1999) found that the trend in increase in extreme high water levels at Buenos Aires from 1905 – 1993 is a result of the same influences causing a trend in relative mean sea-level rise. Woodworth and Blackman (2002) found no significant increase in extreme high water levels from Liverpool from 1768 – 1993 above that which is explained from effects of tidal amplitudes, mean sea-level, and vertical land movement. Woodworth and Blackman (2004) analyzed sea-level data from 141 globally distributed tide gauges and found evidence of an increase in extreme high water levels since 1975 paralleling changes in regional climate as well as short-term variability in mean sea-level, indicating that the factors that produce change in mean sea-level also cause change in extreme high water levels to the same degree, and additional factors, such as variations in storminess, are not a large forcing agent explaining the change in extreme waters. Analysis of tide gauge data from Tuvalu found evidence that extreme water events are most likely a result of the same atmospheric and oceanic factors as cause change in mean relative sea-level (Hunter, 2002).

Conversely, extreme water events in Stockholm in the Baltic Sea have been found to have a greater response to wintertime North Atlantic Oscillation forcings (fluctuations in air pressure and wind fields) than does mean sea-level (Woodworth and Blackman, 2004). Church et al. (2004b) found a rise in elevation of extreme high water events at the 99.9 percentile to be only slightly larger than the rise in median sea-level for Fremantle, Australia (similar to the results from American Samoa), but found the rise to

be significantly larger than the rise in median sea-level for Fort Denison, Australia, indicating some influence other than that causing the rise in sea-level at the latter location. Bijl et al. (1999) found no discernible trend over the last century of non-tidal sea-level variability around the UK and eastern North Sea above the large natural decadal sea-level variability.

### **5.2.2. Change in Frequency of Extreme High Water Events**

Analyses of the Pago Pago tide gauge data indicate a lack of change in the annual number and annual return period of extreme high water events of a given height above each year's median sea-level over the observed 55 year time series (Figs. 4.3 – 4.6). This suggests that factors causing extreme waters other than the atmospheric and oceanic influences that are causing global sea-level to rise, such as storminess from variations in regional climate, have not been changing in frequency in the vicinity of American Samoa over the observed time period. This is not consistent with the regional observation made by Church et al. (2006), that there has been a significant increase in monthly sea-level variance after 1970 in the Pacific region, indicating that there has been an increasing trend in inter-decadal variability, such as from ENSO events. This observation implies that, while not detected in American Samoa, regionally, an increase in the frequency of extreme events has been occurring, and may become an increasingly prominent management issue.

Analysis of return period of both 99.9 and 99.95-percentile extreme high water events indicate little change in frequency over time. The confidence interval around the point estimates for the length of return periods of 99.9 and 99.95-percentile extreme high water events both include zero slope indicating no linear trend (Figs. 4.4 and 4.6). The midpoint of the data series falls in mid-1976. There may have been a sudden climate shift at this time: Graham (1995) observed a sharp rise in global average tropospheric temperatures since the mid-1970s, likely a response to increased atmospheric carbon dioxide. There was no evidence of an effect of this climate shift on extreme high water event frequency. Average return time of 99.95-percentile extreme high water events decreased 7% from the first to second half of the observed 55-year period while the average return time of

99.9-percentile extreme high water events increased 17% (Fig. 4.7). These observed changes in length of return period are both small, where the difference in results may be a result of the definition employed of an extreme high water event, which resulted in small data series ( $N = 81$  and  $121$  for the 99.95 and 99.9-percentile return period analyses, respectively), such that the trends in frequency (number of events per year) of both 99.9 and 99.95-percentile extreme high water events both had confidence intervals that overlapped 0 indicating no linear trend (Fig. 4.4 and 4.6).

There have been no previous studies assessing site specific change in frequency of extreme high water events of a given height above each year's median or mean relative sea-level. Church et al. (2004b) found a decrease in return period of extreme high water events above a given height above gauge datum (a different definition of an extreme high water event than employed in this study) by factors of between 2 and 3 for two locations in Australia (Fremantle and Fort Denison) when comparing periods pre and post 1950. This confirms that the level of extreme high waters is increasing at these sites, which is expected at sites experiencing relative sea-level rise.

### **5.2.3. Regional Implications**

American Samoa is in a region that experiences large El Nino - Southern Oscillation (ENSO)-related steric changes lasting several months to years (Fig. 4.1, and see discussion in Section 5.1). As a result, in the tropical Pacific, extreme high waters are likely to be related to mean sea-levels (Hunter, 2002; Woodworth and Blackman, 2004). For example, annual mean sea level in the tropical Pacific can change by tens of centimeters on inter-annual to inter-decadal scales due to the ENSO cycle (Church et al., 2004a, 2006): annual mean sea-level at some locations can change by as much as 20– 30 cm on inter-annual time scales (Church et al., 2006), and a maximum response to the 1986-1987 ENSO event, which was exceptional in magnitude, occurred near Papua New Guinea in the western Pacific, where steric sea level varied by 56 cm (Ridgway et al., 1993). As discussed in Section 5.2.2, long-term tide gauge records from the tropical Pacific document that the variance of monthly mean sea-level after 1970 was about double that before 1970, consistent with the observed trend of more frequent,



persistent and intense ENSO events over the past two decades (Folland et al., 2001; Church et al., 2006). This increase in variance in monthly mean sea-levels is expected to result in increased frequency of extreme high water events at a given level (Church et al., 2006). However, not all regions experience large ENSO-related steric sea-level changes (Church et al., 2004b; Woodworth and Blackman, 2004). There is spatial variability in trends in elevations and frequency of extreme high water events, warranting site-specific analysis as conducted here for American Samoa.

#### **5.2.4. Method for Analysis of Extreme High Water Events**

A study of annual sea-level maxima can result in large error if even a small number of hourly values were incorrectly measured around the high water. Focus on the 99.9 and 99.95 percentile levels provides the information on extremes needed for this analysis, but are relatively susceptible to error from inaccurate data (Woodworth and Blackman, 2004). For example, there were small sample sizes in this study for some study components (N = 81 and 121 for the 99.95 and 99.9-percentile return period analyses, respectively, and N = 55 for both the 99.95 and 99.9-percentile frequency analyses). Errors from tide gauge malfunction or other source that affected even a small number of these data series could potentially have altered results of these analyses.

### **5.3. HISTORICAL RECONSTRUCTION OF MANGROVE MARGIN POSITIONS**

Determining trends in horizontal position of mangrove margins provides one component of information needed to predict mangrove response to any future sea-level rise (Sections 1.5.6, 1.5.7). The decadal change in position of three American Samoa mangroves was predicted from an analysis of a time series of remotely sensed imagery, GIS techniques, and projections for change in sea level relative to the mangrove surface. The observed mean landward migration of the mangroves' seaward margins over four decades was 25, 64, and 72 mm a<sup>-1</sup>, 12 to 37 times the observed regional relative sea-level rise rate. The observed change in position of the seaward margins of Masefau and Nu'uuli mangroves, over years and decades between

observations, has been a highly significant ( $P < 0.01$ ) linear temporal trend: there have been clear linear trends in reductions in mangrove area resulting from movement of the seaward margins (Fig. 4.15). Leone's seaward position has been highly variable with no significant linear temporal trend (Fig. 4.15). Over these periods of several years to decades between observations, the highly significant linear temporal trend at Maseafu and Nu'uuli, and highly significant correlation with the trend in relative sea-level rise, suggest the primary cause of migration is from rising sea-level, discussed in Section 5.6. Filling within Leone mangrove for development was likely the primary cause of variable nonlinear temporal position of the seaward margin, which would explain the lack of correlation with relative sea-level trends.

If there were a way to reconstruct the position of the mangrove landward margins over recent decades to observe a trend in horizontal movement of this boundary at accuracies of a centimeter or better per year, as conducted for the seaward mangrove margins in this study, this would account for all factors affecting the position of the mangrove landward margin, and would provide an accurate way to predict future movement. However, at many mangrove sites, including the three sites from American Samoa, it is not possible to accurately identify the landward mangrove margin from interpretation of historical aerial photos and satellite imagery (e.g., Woodroffe, 1995; Solomon et al., 1997; El-Raey et al., 1999; Wilton and Saintilan, 2000; Saintilan and Wilton, 2001). Improvements in remotely sensed imagery, and techniques for interpretation, to separate mangrove from non-mangrove forest cover at a precision and accuracy to detect changes in position of cm per year or better, could increase the precision and accuracy of assessments of mangrove vulnerability to relative sea level rise. This may be achieved by the growing field of active remote sensing, such as Synthetic Aperture Radar C-band or L-band sensors, to identify the location of mangrove boundary position and extent as well as to discriminate between mangrove species and zones, and potentially identify biomass (Lucas et al., In Press). To produce an accurate observation of the rate of change in the position of the landward mangrove margin using this technique will require a

time series of imagery enabling this precise level of interpretation dating back several decades.

Two assumptions made in estimating future landward migration of mangrove landward margins were that no new development will occur to obstruct the migration of the mangrove, and that existing development will remain in place and be fortified, and not be moved or abandoned. In American Samoa, only the second assumption is realistic. As a result, the scenarios for changes in mangrove area as a result of movement of mangrove margins are likely conservative underestimates for future losses of mangrove area: Additional development next to mangroves will likely occur over coming decades, and existing and new development will likely be protected from erosion and inundation.

There is little quantitative information available on land use changes in American Samoa's watersheds. Williams (2004) quantified land uses in 1961, 1984, and 2001 for the Tafuna Plain, Tutuila Island, American Samoa, an area adjacent to the Nu'uuli mangrove study site. Results showed that, over the four decades, the area of forested land decreased by 52%, while the area of developed land increased 367% (Williams, 2004). This may have altered sediment, freshwater, and pollutant input levels into mangroves, causing change in position of margins as well as affecting health, resistance and resilience to rising sea-level. Qualitative analysis of a time series of images of the three mangrove study sites revealed direct losses of mangrove area from filling, such as placement of fill within the Leone mangrove, as well as activities that likely altered mangrove functions, such as development of the Pago Pago airport runways across Pala Lagoon, dredging to create the runways, and increasing development of the Tafuna coastal plain within the Nu'uuli mangrove watershed contributing area. These activities may have reduced the lagoon water turnover rate, altered the tidal range, increased the sedimentation rate in the lagoon, and altered the sediment, nutrient, freshwater and pollutant input levels into the mangrove.

Uncertainties reported in Table 4.1 for observed changes in mangrove areas resulting from changes in position of the mangrove seaward margins, based on the co-registration error from IKONOS reference image (reported in Appendix 2), were generally small: In ten cases the uncertainty was less than

3.6% of the total observed change in mangrove area, and in the remaining two cases, the uncertainty was 13 and 17% of the total observed change in mangrove area. Some uncertainty is also introduced from human error in digitizing the mangrove boundary line, which is not accounted for in this reported estimate. Some of the observed areas of smaller polygons in co-registered aerial photos used in the analysis are within the co-registration root-mean-square (RMS) error (Appendix 2), reducing confidence in estimates of trends in mangrove area. However, a comparison of coordinates of fixed features between the co-registered aerial photos and the IKONOS imagery, such as corners of buildings, road intersections, and bridges, located adjacent to the boundaries of the mangrove sites indicates that the error is generally small, within a few decimeters to meters. Interpretation of some of the images to identify the mangrove seaward margin was difficult due in part to the poor image contrast and resolution.

The observed mean landward migration of three mangroves' seaward margins represents the first reported landward rate of migration of seaward mangrove margins at sites where there is a significant positive correlation between the rate of margin migration and rate of relative sea level rise. The next section discusses why predictive models produce inaccurate estimates of mangrove erosion rates caused by rising sea-level.

#### **5.4. BRUUN RULE**

Calculation of the mangrove margin erosion rate using a predictive model of beach erosion called the Bruun rule (Bruun, 1962, 1988), or a modified Bruun rule, was not employed for this study. Mangroves are not expected to respond in accordance with Bruun rule assumptions, and the Bruun rule, as with other general predictive models of coastal erosion, is not suitable for small-scale, site-specific estimates (Bruun, 1988; List et al., 1997; Komar, 1998; Pilkey and Cooper, 2004). The Bruun rule and modifications of the model have been used broadly to estimate erosion of various coastal types despite a large body of evidence that this largely results in inaccurate estimates of both past erosion rates and future erosion estimates at specific

locations and over short time periods (SCOR Working Group, 1991; List et al., 1997; El-Raey et al., 1999; Pilkey and Cooper, 2004).

Bruun (1962, 1988) provides a simplistic model of change to beach profile with sea-level rise. It assumes a closed material balance system so that the migrating beach has no net loss of sand volume, and that there is a uniform sandy shoreface with no outcrops or other obstacles that could cause spatially or temporally non-uniform retreat rates to sea-level rise (Bruun, 1988; Komar, 1998; Pilkey and Cooper, 2004). The Bruun (1962, 1988) model assumes that with increased sea-level, the equilibrium beach profile and shallow offshore migrates upward and landward, the upper beach is eroded due to the landward translation of the profile, the material eroded from the upper beach is deposited immediately offshore, and the rise in nearshore bottom equals the rise in sea-level.

The results from this study observed erosion rates of the mangrove seaward margin of 12.4, 32.2, and 32.6 times the rate of relative sea-level rise for Leone, Masefau, and Nu'uuli mangrove study sites, respectively.<sup>1</sup> This is the first reported landward rates of migration of seaward mangrove margins, in this case at two sites where there has been a significant positive correlation between the rate of margin migration and rate of relative sea level rise. Assuming an average slope of many coastlines of 1-2%, according to the Bruun model, the landward recession rate will be between 50 to 100 times the rate of relative sea-level rise (Bruun, 1962, 1988; SCOR Working Group, 1991; Komar, 1998). The slopes of the three mangrove study sites are estimated to be 0.7% (range 0.2 – 2.5%) based on the local tidal range (~1 m), mean width of the three mangrove study sites of about 75 m (range 2-300 m), and that mangroves are generally located between the level of mean high water spring tides (just above the high tide line) and mean sea level (Ellison, 2001, 2004), the upper half of the tidal range, in this case being 0 - 0.5 m above msl. In this case, the Bruun model predicts an erosion rate of about 143 times the relative sea-level rise rate, which is inconsistent with observations.

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<sup>1</sup> Leone mangrove experienced a 24.5 mm a<sup>-1</sup> landward migration rate of its seaward margin while regional relative sea-level rose at 1.97 mm a<sup>-1</sup>. Masefau mangrove experienced a 63.9 mm a<sup>-1</sup> landward migration rate of its seaward margin while site specific relative sea-level rose at 1.97 mm a<sup>-1</sup>. Nu'uuli mangrove experienced a 72.3 mm a<sup>-1</sup> landward migration rate of its seaward margin while site specific relative sea-level rose at 2.22 mm a<sup>-1</sup>.

Because mangroves have different sediment budget processes than beaches, mangroves are not expected to respond in accordance with Bruun rule assumptions. For instance, mangrove sediment is generally finer grained than that of beaches, and wave energy dissipates as waves progress through the mangroves. Furthermore, the Bruun Rule produces inaccurate erosion estimates because other factors can have considerably larger influence in causing shoreline changes than relative sea-level rise, especially over relatively small temporal and spatial scales (i.e., < 2 years, < 1 km<sup>2</sup>) (Bruun, 1988; List et al., 1997; Komar, 1998; Nunn, 2000; Donnelly and Bertness, 2001; Pilkey and Cooper, 2004).

Until reliable predictive elevation models are developed for mangrove ecosystems (Section 8.2), site-specific monitoring is necessary to assess mangrove vulnerability and predict responses to sea-level rise. Now that results from a historical reconstruction of mangrove seaward position have been discussed, including a discussion of the state of predictive elevation models for mangroves, a discussion of site-specific assessment of vulnerability to relative sea-level rise is continued, based on observed trends in mangrove sediment surface elevation and the rate of regional relative sea-level rise.

## **5.5. TRENDS IN MANGROVE SURFACE ELEVATION AND SEDIMENT ACCRETION RELATIVE TO SEA-LEVEL**

Based on observations of spatial and temporal trends in mangrove surface elevation, this section discusses whether or not results indicate the American Samoa mangrove study sites have been keeping pace with relative sea-level rise. This assesses one aspect of mangrove resistance to sea-level rise (Section 1.1) by determining a mangrove site's past ability to keep pace with rising sea-level and maintain vs. alter its current position, with implications for whether or not the ecosystem will experience alteration to its functions, processes and structure (Odum, 1989; Bennett et al., 2005). The section also describes the contribution to elevation trends from near surface versus deeper subsurface factors, and compares these results to those reported from previous studies. The stakes and <sup>137</sup>Cs and excess <sup>210</sup>Pb activity dating

methods are critiqued for their utility in assessing mangrove vulnerability to sea-level rise. Results related to spatial patterns and temporal cycles in elevation trends are discussed. Finally, results from American Samoa and the general state of knowledge of the relationship between geomorphic setting and change in sediment surface elevation are discussed.

### 5.5.1. Mangrove Sedimentation and Surface Elevation Trends

Sea-level has been rising at  $2.22 \text{ mm a}^{-1}$  ( $\pm 2.22$  95% CI) relative to the sediment surface of Nu'uuli mangrove, indicating that the mangrove has likely been experiencing site-specific relative sea-level rise. The Nu'uuli site specific relative sea-level rise rate is the difference between the rate of change in regional relative sea-level and rate of change in sediment surface elevation. The regional relative sea level rise rate of  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI) ( $1.65$  to  $2.29 \text{ mm a}^{-1}$ ) just touches the error interval of the rate of change in elevation of the mangrove surface of  $-0.25 \text{ mm a}^{-1}$  ( $\pm 1.90$  95% CI) ( $-2.15$  to  $1.65 \text{ mm a}^{-1}$ ). Results of observations of the landward movement of Nu'uuli's seaward margin are consistent with there having been a rise in site-specific relative sea-level over the past four decades. Nu'uuli will not keep pace with projected relative sea-level rise scenarios that include an increase in the current relative sea-level rise rate in American Samoa, including the upper projection equivalent to a linear rate of  $8.8 \text{ mm a}^{-1}$ .

Fig. 1.8 illustrates influences of the shallow, active root zone, middle mangrove peat zone, and lower portion of the soil profile on the elevation of a mangrove sediment surface. In Nu'uuli and Masefau mangroves, estimates of change in sediment surface elevation accounting for the full soil profile are based on observations from elevation stakes. The boundary between the upper and lower soil profile is based on observations of the depth of peak  $^{137}\text{Cs}$  activity and lower depth of excess  $^{210}\text{Pb}$  activity in Nu'uuli and Masefau mangroves, respectively.

Subsurface processes below 15.5 cm that affect sediment elevation have contributed about  $-4.05 \text{ mm a}^{-1}$  of elevation lowering in Nu'uuli mangrove: Nu'uuli was observed to experience  $-0.25 \pm 1.90 \text{ mm a}^{-1}$  of lowering of the sediment surface based on observations of the entire soil profile, while there was a  $+3.8 \text{ mm a}^{-1}$  change in elevation in the upper 15.5 cm of the profile, such that  $-0.25 \text{ mm a}^{-1}$  absolute change minus  $3.8 \text{ mm a}^{-1}$  upper 15.5 cm =  $-4.05 \text{ mm a}^{-1}$  rate of lowering of the profile below 15.5 cm

depth. For Nu'uuli mangrove, the rate of change in sediment elevation due to surface processes is based on the observed depth of peak  $^{137}\text{Cs}$  activity at 15.5 cm depth, accounting for about the upper 10% of the mangrove sediment profile. Elevation controls in this upper portion of the sediment profile likely include sediment accretion and erosion, surface root growth, and near-surface mangrove belowground root productivity, while main processes deeper in the sediment profile include organic matter decomposition, subsidence and autocompaction, and fluctuations in water table levels and pore water storage (Lynch et al., 1989; French, 1991; Cahoon et al., 1999; Donnelly and Bertness, 2001; Krauss et al., 2003; Rogers et al., 2005; Cahoon and Hensel, 2006).

Masefau mangrove has been experiencing site-specific relative sea-level rise of  $1.97 \text{ mm a}^{-1}$  ( $\pm 0.32$  95% CI). Masefau mangrove has been experiencing site-specific relative sea-level rise, based on there being a significant difference between the rate of change in regional relative sea-level and observation of no linear trend ( $0 \text{ mm a}^{-1}$ ) in sediment surface elevation: This would have caused the mangrove to retreat landward over past decades. Observed landward movement of Masefau's seaward margin over the past four decades is consistent with there having been a rise in site-specific relative sea-level over this same period.

Subsurface processes below 27 cm in the soil profile that affect sediment elevation have contributed about  $-3.3 \text{ mm a}^{-1}$  of elevation lowering in Masefau mangrove (no absolute change in sediment elevation, or  $0 \text{ mm a}^{-1}$  rate of change in elevation of the sediment surface accounting for the full soil profile minus the upper 27 cm of the soil profile change in elevation rate of  $+3.3 \pm 0.3 \text{ mm a}^{-1}$ ). For Masefau mangrove, the rate of change in elevation due to surface processes is based on the observed lower depth of excess  $^{210}\text{Pb}$  activity at 27 cm depth, accounting for about the upper 20% of the mangrove sediment profile.

In summary, there was a rise in sediment elevation of  $+3.8$  and  $+3.3 \text{ mm a}^{-1}$  in Nu'uuli and Masefau, respectively from sediment accretion and shallow subsurface processes, a  $-4.05$  and  $-3.3 \text{ mm a}^{-1}$  lowering in sediment elevation from deeper subsurface processes in Nu'uuli and Masefau, respectively, with a net elevation change rate of  $-0.25 \text{ mm a}^{-1}$  in Nu'uuli and



no rate of change ( $0 \text{ mm a}^{-1}$ ) in Masefau. These results are consistent with previous studies, which document large differences between trends in sediment accretion and trends in sediment elevation in mangroves (Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon et al. 2006). The large contribution of deeper subsurface controls on elevation in Nu'uuli and Masefau mangroves is consistent with observations of substantial control of mangrove sediment elevation from groundwater changes primarily from the deepest soil horizon adjacent to bedrock (Whelan et al., 2005).

Ellison (2001) found a  $1.45 \text{ mm a}^{-1}$  mean accretion rate in Nu'uuli mangrove over the past 620 ( $\pm 70$ ) years, and a  $3.46 \text{ mm a}^{-1}$  accretion rate in Masefau mangrove over the past 410 ( $\pm 60$ ) years, based on radiocarbon dates of basal organic mud samples. The rate of change in sediment elevation from surface and shallow subsurface processes for Nu'uuli mangrove observed in this study over the past four decades, based on analyses of sediment cores, is more than twice the sediment accretion rate over the past several hundred years reported by Ellison (2001). This difference might be due to increased upland erosion from increasing land use activities in the mangrove's contributing watershed, with concomitant increased supply of terrigenous sediment to the mangrove (Williams, 2004). The lack of significant change for Masefau between the observed rate of change in sediment elevation from surface and shallow subsurface processes over the past eight decades, based on analyses of sediment cores, versus over the past several hundred years as reported by Ellison (2001), may be a result of a lack of a substantial increase in development activities in the wetland's watershed. Nu'uuli mangrove is within a rapidly developing catchment with extensive clearance and development adjacent to the mangroves in the last couple of decades, while Masefau mangrove is within a relative undeveloped catchment.

Based on an assessment of stratigraphic records of Pacific Island mangroves during sea-level changes of the Holocene Period, Ellison and Stoddart (1991) generalized that low island (atoll) mangroves, which generally accumulate vegetative detritus for substrate build-up and lack a large source of inorganic sediment, can keep up with a  $1.2 \text{ mm a}^{-1}$  relative sea-level rise rate. They further generalize that mangroves of high islands

and continental coastlines, which have relatively large supplies of terrigenous inorganic and organic sediment from rivers and longshore drift, can keep pace with a  $4.5 \text{ mm a}^{-1}$  relative sea-level rise rate (Ellison and Stoddart, 1991). Mangroves of low relief islands in carbonate settings that lack rivers have been identified in previous studies as likely to be the most sensitive to sea-level rise, owing to their sediment-deficit environments (Thom, 1984; Ellison and Stoddart, 1991; Woodroffe, 1987, 1995, 2002). The observed trends in sediment surface elevation in the two high island mangroves of American Samoa identified elevation thresholds ( $-0.25$  and  $0 \text{ mm a}^{-1}$  for Nu'uuli and Masefau mangroves, respectively) substantially different from and lower than the  $4.5 \text{ mm a}^{-1}$  sedimentation threshold generalized by Ellison and Stoddart (1991). As reviewed in the Introduction (Section 1.5.4), there are numerous surface *and* subsurface controls on mangrove sediment elevation. Recent studies have shown that subsurface controls on mangrove sediment elevation can offset high or low sedimentation rates (Cahoon et al., 2006; Cahoon and Hensel, 2006): High rates of subsidence in sediment-rich settings may result in a trend of elevation lowering while low rates of subsidence in sediment-poor settings may result in rising sediment elevation and lower vulnerability to any rise in relative sea-level. Because large and significant differences between trends in mangrove sediment accretion and sediment elevation have been observed, and in many studies subsurface process controls have been observed to be a larger factor than surface processes in determining sediment elevation linear trends, as demonstrated in this study in Nu'uuli and in previous research (Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon et al. 2006), sedimentation rates taken alone are not a suitable indicator of mangrove vulnerability to change in relative sea-level.

The observations from the two American Samoa mangroves of changes in sediment surface elevation, and trends in surface and shallow subsurface elevation, are within the range of results reported in the literature. Sediment accretion rates have been observed to range from  $+1.0$  to  $+41 \text{ mm a}^{-1}$  (Chapman and Ronaldson, 1958; Bird, 1971, 1980; Cahoon and Lynch, 1997; Rogers et al., 2005a,b; Cahoon et al., 2006; McKee et al., 2007). Sediment elevation rates range from  $-40$  to  $+7.2 \text{ mm a}^{-1}$  (Krauss et al., 2003;

Cahoon and Lynch, 1997; Rogers et al., 2005a,b; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007). Based on a review of 28 mangrove study sites from the Wider Caribbean and Western Pacific regions, Cahoon and Hensel (2006) found that the mean net mangrove sediment elevation change was  $+1 \text{ mm a}^{-1}$ , where vertical accretion averaged  $5 \text{ mm a}^{-1}$  and subsurface processes contributed elevation lowering of  $-4 \text{ mm a}^{-1}$  (estimates of error for these means were not reported) and that elevation trends for the majority of sites are not keeping pace with current rates of regional relative sea-level rise. As the case with the two American Samoa mangroves, in sites where accretion rates exceeded rates of regional relative sea-level rise, Cahoon and Hensel (2006) and Cahoon et al. (2006) found that most of these sites had experienced elevation deficits with respect to regional relative sea-level rise.

The low 10-30% organic content of the Tutuila mangrove muds, consistent with there being high terrestrial sediment inputs, and perhaps indicating that root growth is not substantially exceeding decomposition, is consistent with the observed relatively high sediment accretion rates determined through radioisotope analyses. These results are consistent with the observation of a negative correlation between soil organic matter content and vertical accretion in the mangrove SET-MH network (Cahoon and Hensel, 2006). Cahoon and Hensel (2006) found no positive correlations between sediment surface elevation change and soil properties.

#### **5.5.2. Elevation Stakes Method**

Results indicate that the elevation stakes sampling method is relatively precise and replicable: If a researcher measured the elevations of 10 stakes at a station and repeated the measurements a second time, the two mean measurements would be within about 1.2 mm. Measures taken to maximize consistency in the location where measurements were taken, by taking the elevation measurement at the same direction and distance from the stake and use of a level, likely contributed to maximizing the precision and replicability of the measurement method.

The inclusion of a relatively large number of sampling locations for observing trends in sediment elevation in the sampling design may have

contributed to the observed high degree of variability. Observations of disparate trends in sediment elevation within different vegetation communities of an individual mangrove (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007) support the inference that inclusion of a larger number of sampling locations across different vegetation communities and localized geomorphic settings can result in higher variability and a wider estimate of uncertainty around an average point estimate for the mangrove system's surface elevation trend. This is the first study to employ broad spatial sampling to observe trends in sediment elevation, with 200 individual measurement stakes in Nu'uuli mangrove and 130 stakes in Masefau mangrove. This large number of sampling units and design for locating the sampling stations was employed to ensure that the sites were sufficiently sampled so the resulting data characterize the entire system. In comparison, other studies designed to measure trends in mangrove and salt marsh sediment elevation, most which use SET-MH technology, have included about 12 sampling locations per wetland system (range of 6-27) (Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007).

Observations over decades, or at least several years, may be necessary to discern statistically significant linear, secular trends in the sediment surface elevation of coastal wetlands. The limited study period (ca. 16 and 18 months for Masefau and Nu'uuli, respectively) has also contributed to the observed large uncertainties around the point estimates of trends in change in elevation for Nu'uuli and Masefau mangroves. The large error intervals (e.g., Fig. 4.18a for individual stations) are likely due to the presence of cyclical events that affect the mangrove surface elevation, and the occurrence of episodic events that cause short-term pulses in sediment erosion and accretion (French and Stoddart, 1992; Kirby et al., 1993; Semeniuk, 1994; List et al., 1997; Reed et al., 1999; Krauss et al., 2003). Nonetheless, this uncertainty was explicitly accounted for in the random effects nonparametric modeling approach adopted in this study (e.g., Fig. 4.17). Trends in elevation change would likely be more apparent, while signals from short-term, episodic, cyclical, and small-scale events would be less apparent over a longer time scale of at least several years but may

possibly require decades (SCOR Working Group, 1991; Semeniuk, 1994; List et al., 1997; Pethick, 2001; Pilkey and Cooper, 2004). There is a global need for substantially longer time series in studies monitoring trends in mangrove and salt marsh surface elevations: The mangrove sites where SET-MH technology has been employed, or similar method for which results have been reported, had been monitored for periods of only about 2.6 years, ranging between 1 and 3.6 years (Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007), comparable to the period of observation in this study. Furthermore, as explained previously, because sediment surface elevation patterns are variable between vegetation communities of an individual mangrove site (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007), and because there is the potential for high variability even within a single vegetation community of a site (Fig. 4.18a), large mangrove spatial sampling needs to be incorporated into sampling designs in order to provide results on trends in sediment surface elevation that accurately characterize the entire site. Experiments that employ study designs that adopt this appropriate, broad spatial sampling will likely find that substantially longer study periods than previously estimated (e.g., minimum of three years, Cahoon and Hensel, 2006), are necessary to demonstrate linear, secular trends.

In retrospect, the method employed to compare monitored trends in sediment elevation to regional relative sea-level would have been improved by having established local benchmarks located adjacent to each mangrove site. The locally established benchmarks could be located so that all elevation monitoring sampling locations within the mangrove were within about 1 km from a benchmark. Then, surveying would occur from the local benchmarks to a benchmark located on the American Samoan level/height network. If there were an indication that the mangrove site benchmark is not stable, then surveying would be conducted on a regular basis. Otherwise, the connection could be repeated annually. Surveying between the local benchmarks and the American Samoan height network, which might be over distances of a few kilometers, could be conducted using GPS heightening or geodetic leveling techniques (American Society of Civil Engineers, 2000).

Any observed differences in trends in elevation between the American Samoan height benchmarks and local mangrove benchmarks would be used to correct the trend in relative sea-level for each individual local mangrove benchmark consistent with the American Samoan height definition. The purpose of surveying between the local benchmarks and American Samoan height network would be to establish the stability of the local mangrove site benchmarks, to determine if there is any change in elevation of the local benchmarks relative to the defined American Samoan height datum, which would result in a different rate of change in relative sea-level as to the rate measured by the Pago Pago tide gauge.

A separate but related matter is that it would be useful to confirm that the stability of the benchmark at the tide gauge is periodically verified. Periodic surveys need to be conducted to determine if there is any differential movement, such as crustal tilting, between the tide gauge benchmark and benchmarks located on the American Samoan level/height network. For example, on Tuvalu, Church et al. (2006) found tilting of about  $0.6 \text{ mm a}^{-1}$  over a distance of only 2.5 km. And, for instance, since 6000 BP there is documentation of site-specific variability in relative sea-level, with site-specific differences in the timing and magnitude of the middle to late Holocene high stand and nature of sea-level fall, in the Pacific Islands region and along different sections of coastline of a single island (Clark, 1990; Dickinson and Green, 1998; Goodwin and Grossman, 2003; Woodroffe and Horton, 2005). Relative sea-level as measured by the Pago Pago tide gauge may be substantially different from trends in sea-level at mangrove sites due primarily to changes in elevation of the mangrove sediment surface (this study; Cahoon and Lynch, 1997; Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007), but also due to slight differences in local tectonic processes, coastal subsidence, sediment budgets, and meteorological and oceanographic factors (e.g., changes to wind, currents and wave energy and direction). In other words, the sections of coastlines where the mangrove study sites are located may be undergoing slightly different relative sea-level trends than the area of Tutuila Island where the Pago Pago tide gauge is located.

The connection between the local mangrove benchmarks and the American Samoan level network benchmarks would allow relating the elevation of the local benchmarks to a mean sea level datum. It would also be possible to survey between the local mangrove benchmarks and each individual stake at each monitoring date in order to verify that the top of each stake does in fact remain at the same elevation over time relative to the local benchmark, as has been assumed in the large body of literature that has employed stakes (Section 1.5.6) (Lee and Partridge, 1983; Kirby et al., 1993; Bradbury et al., 1995). A local mangrove benchmark would presumably be more stable than the stakes, as the stakes are subject to disturbance from debris, people, and perhaps natural factors, such as variability in soil pore pressure. Also, while not necessary for the purpose of determining mangrove vulnerability to relative sea-level rise, this would make it possible to relate the elevations of the mangrove surface at each individual monitoring stake to the common mean sea-level datum by surveying between each individual stake to the local benchmarks through the level network connection to the tide gauge. However, surveying via leveling in mangroves would be extremely difficult given the high density of trees and the soft sediment, possibly requiring specially-designed tripods to eliminate or correct for sinking into the sediment.

Employing electronic distance meter (EDM) leveling methods (American Society of Civil Engineers, 2000) in mangroves is challenging because the soft sediment surface, and in some locations, dense undergrowth, reducing accuracy. Measurement precision for heights should be within a range of 5-10 mm over a mangrove site (personal communication, 1 November 2007, Prof. Richard Coleman, University of Tasmania).

Use of GPS technology to measure elevation at a precision and accuracy to allow derivation of rates of change in millimeters per year could increase the practicality of assessments of mangrove vulnerability to relative sea level rise, as well as improve accuracy by better characterizing mangrove sites by making it more practical to include a larger sample size. However, current GPS technology cannot be used under dense forest canopy to provide precise and accurate vertical measurements due to data

dropout from loss of satellite lock. Improvements in differential GPS technology that enable achieving millimeter precision and accuracy for vertical measurements of the mangrove substrate over distances of a few kilometers within mangrove forests, with dense canopy cover, would enable the use of GPS to monitor trends in the elevation of mangrove substrate at specific points. Compared to other methods to measure changes in elevation of mangrove surfaces, such as the use of stakes in this study, and SET technology (Cahoon et al., 2002; Cahoon and Hensel, 2006), if GPS technology were developed for this purpose, it would enable the inclusion of a far greater number of sample points in individual mangrove study sites to better characterize the site.

### **5.5.3. Sediment Core Radionuclide Analyses Methods**

Cores taken for radionuclide analyses to estimate accretion rates are subject to error if they are taken from locations subject to a source of sediment mixing or compaction from biotic and abiotic processes, such as an area with significant crab bioturbation, next to a tidal creek, stream, river, trail, piggeries, or agricultural plots (e.g., Lynch et al., 1989). In the Nu'uuli core,  $^{137}\text{Cs}$  activities at depth  $\geq 25$  cm are considerable and more prominent than in the region 0-17 cm. If the suggested accretion rate of  $3.8 \text{ mm a}^{-1}$  is valid, this means that the  $^{137}\text{Cs}$  presence at depths below 25 cm was a result of disturbance before core extraction. Alternatively, if profile disturbance had not occurred, the detection of  $^{137}\text{Cs}$  at 37-38 cm depth, and likelihood of continued presence below this depth, produces an average sediment accretion rate of  $> 7.5 \text{ mm a}^{-1}$  over the 50-year period from the year of first significant  $^{137}\text{Cs}$  fallout (1954) to the date of core extraction, which conflicts with the rate based on the apparent depth of peak deposition. It is very likely that there has been vertical sediment movement from bioturbation, such as by crabs, or desorption and diffusion moving  $^{137}\text{Cs}$  to deeper sections of the sediment profile.

Furthermore, because  $^{137}\text{Cs}$  fallout was particularly low in the band of 10-20° S. latitude (Agudo, 1998), which includes the study area, while  $^{137}\text{Cs}$  activity levels were within the detection limits of the Ge detector employed for the radioisotope analyses, the relatively low concentration of  $^{137}\text{Cs}$  in the



sediment profile may have contributed to the variability observed in the  $^{137}\text{Cs}$  activity depth profile. If the accretion rate from the apparent peak deposition is correct, then the method would have been improved by using sections with a thickness of  $\leq 4$  mm to increase the resolution in the  $^{137}\text{Cs}$  vs. depth curve. This would have provided 3-4 points at the peak concentration instead of just one point found within the 1-cm thick sections. However, this might not have produced a large enough soil sample per section as required for analysis. Furthermore, for both the Nu'uuli and Masefau cores, the results derived from the  $^{137}\text{Cs}$  data and the  $^{210}\text{Pb}$  data are not consistent. It would be useful to collect additional cores at the two study sites, again, from locations where the likelihood of disturbance having occurred is low, to attempt to produce results where the  $^{137}\text{Cs}$  and  $^{210}\text{Pb}$  conclusions are consistent. However, because  $^{137}\text{Cs}$  activities are relatively low, for instance in the Nu'uuli core 9 of the 17 sections show zero activity, this technique may be unsuitable for analysis of cores from this region. While the Nu'uuli  $^{137}\text{Cs}$  results do show a discernible peak and allows a good correspondence with the sedimentation rate derived from the  $^{210}\text{Pb}$  profile at Masefau, due to the high variability in the results, there is strong evidence of contamination to both cores, creating low confidence in the resulting accretion estimates.

The sedimentation rates estimated from measurement of  $^{137}\text{Cs}$  and excess  $^{210}\text{Pb}$  activity from a single core from each study site was very likely inadequate to characterize the entire site because large spatial variability in sedimentation rates have been observed across individual mangroves (e.g., Krauss et al., 2003). However, at a cost of about USD \$1500 per core for laboratory analyses alone, this method is not economical, especially for the less developed countries of the Pacific Islands region (Table 6.1). Furthermore, results from these methods provide limited information on mangrove vulnerability to sea-level rise, as they do not account for all process controls on sediment elevation. Employment of marker horizon method in combination with stakes inserted through the sediment profile or the SET method would have been a better approach to provide information to separate the contributions of surface and subsurface factors on sediment elevation (Cahoon et al., 2006).

#### **5.5.4. Spatial Patterns and Temporal Cycles in Elevation Trends**

Vanderzee (1988), using models for salt marshes with sediment supply not quite sufficient to keep pace with sea level rise, predicted that the seaward margins will erode and be deposited in the mid and upper marsh to contribute to accretion. None of the Nu'uuli or Masefau transects showed this pattern: There were no statistically significant differences in sediment surface elevation trends between stations within sites, although the sampling transect effect was highly uncertain (Fig. 4.18a).

There was no cyclical or seasonal trend in sediment surface elevation at Nu'uuli, where a declining linear trend was the best fit (Fig. 4.17a). There was some evidence of a cyclical trend, and perhaps seasonality, at Masefau over the 1.3 year period, but with low confidence, where a substantially longer time series is likely needed to confirm any cycles (Fig. 4.17b). Large pulses of sediment accretion might be expected during the American Samoa winter rainy season (inconsistent with Fig. 4.17b) or individual storm events, when surface runoff from the mangroves' catchment carries terrigenous sediment into the mangroves and groundwater levels are raised. There might also be pulses of sediment erosion during extreme high water events or during seasons/periods with higher wind and wave energy. For instance, Kirby et al. (1993) found a unidirectional trend of long-term accretion and erosion in portions of their study site in Northern Ireland, but also observed a seasonal cycle of winter and spring erosion and summer accretion, hypothesizing that wind-generated waves caused erosion in winter and spring, and summer accretion was enhanced by calmer weather and algal binding, with is generally consistent with Fig. 4.17b. Krauss et al. (2003) observed pulses in elevation increase over a 2.5-year study period in Pacific island mangroves using stakes, hypothesizing that the pulses are caused by episodic rainfall and bioturbation events.

#### **5.5.5. Geomorphic Setting as an Indicator of Mangrove Vulnerability to Sea-Level Rise**

Results indicate that, using an explicit factor for study site, there was a highly significant difference in the mean sediment elevation change between the two American Samoa mangrove study sites. Sediment elevation change was

significantly lower at the embayment fringe site (Nu'uuli) compared to the estuarine basin site (Masefau). The Nu'uuli embayment site change in sediment surface elevation for site effect from the VCGAMM (Fig. 4.18b) was an estimated  $-3.432 \text{ mm a}^{-1} \pm 1.267 \text{ SE}$  relative to a 0 reference level for Masefau, the estuarine basin site. It was hypothesized that mangroves in an estuarine geomorphic setting would be more resistant to regional relative sea-level rise than those in an embayment setting because estuarine mangroves are likely to have a higher rate of change in the sediment surface elevation due to having greater hydrologic turnover, larger freshwater inputs and lower salinity, higher primary productivity, and higher sedimentation rates than embayment mangroves (Thom, 1982, 1984; Twiley, 1988; Woodroffe, 1987, 2002). This is the first reported documentation of significantly different mean sediment surface elevation change for mangroves in different geomorphic settings.

However, there has been a lack of an observed correlation between geomorphic class and change in mangrove sediment elevation from an ad hoc global database: Cahoon and Hensel (2006) found no significant correlations between geomorphic class and sediment elevation trends, based on a review of 28 mangrove study sites employing the SET-MH method, from the Wider Caribbean and Western Pacific regions. Given this state of knowledge of relationship between geomorphic setting and change in sediment surface elevation, until reliable predictive elevation models are developed for mangrove ecosystems, site-specific monitoring is necessary to assess vulnerability to sea-level rise.

The hypothesis that estuarine mangroves will be more resistant to relative sea-level rise than embayment mangroves (Section 1.6, Thesis Aim 1, Hypothesis 1) is based on a broad generalization, and may not be consistently observed at other locations. This is because a site-based geomorphic setting classification is based on the primary overall setting characteristics (Section 1.5.1). But an individual site can contain numerous categories of settings within the site. Woodroffe (1987, 1992, 2002) recognized that these broad geomorphic categories are not mutually exclusive, such that an individual system might fall between two or more individual geomorphological settings, and might occur in a combination of

settings. For instance, Nu'uuli mangrove as a single unit is best categorized as an embayment mangrove, but contains sub-areas which might be categorized differently than the site as a whole. For instance, a small area of the site is best categorized as a back barrier dwarf mangrove (Section 2.3). As a result of this spatial variability in geomorphic settings within mangrove sites, there is likely to be high variability in hydrologic turnover, sedimentation rates, and other environmental parameters across mangroves categorized in the same geomorphic class.

## **5.6. RELATIVE SEA-LEVEL CONTROL ON MANGROVE MARGIN MOVEMENT**

Sections 5.3 and 5.5 discussed the implications of results from the assessment of whether or not the American Samoa mangrove study sites have been keeping pace with rising sea-level over past decades. This prepares us to now discuss predicted resistance and resilience of the American Samoa mangrove sites to projected relative sea-level rise, starting by discussing the determination of whether or not sea-level rise has been a primary control on mangrove horizontal position.

A determination for individual mangrove sites if a significant correlation exists between the observed rate of change in position of the mangrove margin and rate of change in relative sea level is needed to determine if there is a basis for using projected rates of change in relative sea level to estimate the future change in mangrove position (Saintilan and Wilton, 2001; Wilton, 2002). For sites where this correlation is not significant, it is likely that factors other than change in relative sea-level are predominant in causing past and predicted future change in mangrove position. Where this is the case, the suitable approach to predict the mangrove's future position is to extrapolate from the observed rate of margin movement (Saintilan and Wilton, 2001), as conducted for Leone mangrove. Fig. 4.15 shows that, for Masefau and Nu'uuli mangroves, the observed change in position of the seaward margin, over years and decades between observations, has been a significant linear temporal trend with only minor linear variability ( $R^2$  values from fit of a linear regression model are close to 1). For Leone, the change in position has

been extremely variable with no significant linear temporal trend, with large nonlinear variability in landward and seaward movement (Fig. 4.15). Over the periods of several years to decades between observations, the highly significant linear temporal trend at Maseafu and Nu'uuli, and highly significant correlation with the trend in relative sea-level rise, suggests the primary cause of migration has been rising sea-level. As discussed in Section 5.3, filling within Leone mangrove for development was likely the primary cause of changing position of the seaward margin, and likely explains the lack of correlation with relative sea-level. Relative to Leone mangrove, based on contemporary site visits and historical imagery (Section 4.3), the Nu'uuli and Masefau mangrove sites have experienced little anthropogenic disturbance to their seaward margins. Unfortunately, it was not possible to identify the dates of occurrence of the filling in Leone from the historical imagery. This information would have enabled a determination of whether or not the dates of filling correspond with the observed pattern of change in mangrove area, shown in Fig. 4.15.

The proportion of the landward mangrove margin that is bordered by development proved to not be a useful predictor of whether or not trends in relative sea-level have been a primary control on position of the mangrove seaward margin. In the two mangrove sites where there was a significant correlation between mangrove area and change in relative sea-level, 16.5% (Masefau) and 68% (Nu'uuli) of the landward mangrove margins were obstructed, while in the mangrove not exhibiting relative sea-level as a primary control on mangrove position, 23.4% (Leone) of the landward margin was bordered by development. A sample size of three is too small to base a definitive conclusion, but results from this study suggest that the proportion of obstruction of the landward margin is not a good indication of whether or not change in relative sea-level has been a primary control on mangrove position.

For this analysis, mangrove area at different points in the time series was determined through analysis of movement of only the seaward mangrove margin, due to a lack of available information on historical position of the landward margin. This method may be inadequate for determining if change in relative sea-level has been a primary control on mangrove area and

position because anthropogenic activities may alter the natural landward mangrove margin position but cause only nominal effect on the seaward margin.

Alternatively, it is possible that a high degree of alteration to a mangrove's catchment may significantly alter the natural position of the mangrove seaward margin, but the degree of development along a mangrove landward margin is not a good indicator of degree of anthropogenic activities that affect position of the seaward mangrove margin. Observations of large mangrove filling for conversion to upland for development within Leone mangrove, based on qualitative interpretation of aerial photographs, resulted in loss of a substantial proportion of the mangrove and alteration to a large proportion of the mangrove's seaward margin. This substantial alteration to the Leone mangrove seaward margin occurred with a small proportion of development along the landward margin relative to Nu'uuli mangrove.

Furthermore, it is unclear whether or not anthropogenic activities, other than direct mangrove filling, characteristic of more urbanized catchments, result in a smaller or larger degree of alteration to mangrove area and boundary position than activities characteristic of more rural catchments. It is possible that alterations to sediment and nutrient inputs, two large controls on long-term trends in mangrove sediment elevation and boundary position (Cahoon et al., 2006; McKee et al., 2007), are larger in catchments with a higher degree of agricultural activity versus areas with high residential, commercial and industrial development. Human activities in urban areas that might affect mangrove area as well as health include deforestation, creation of impervious surfaces and concomitant reduced groundwater recharge and increased surface water runoff, groundwater extraction, surface water diversion, stream channelization, and pollutant discharge. Relevant activities characteristic of more rural areas might include clearing of forests and freshwater wetlands for agricultural use, discharge of nutrients and pollutants from farms and piggeries, and soil erosion from unimproved roads.

Three methods were employed to calculate the cumulative change in relative sea-level, as part of the analysis of relative sea-level control on mangrove margin movement (Section 3.6). All three methods resulted in significant correlations for Nu'uuli and Masefau mangroves, and a lack of a

significant correlation for Leone. However, due to large uncertainty in Nu'uuli and Masefau mangroves' trends in sediment surface elevation (Section 4.4, Fig. 4.17) and concomitant uncertainty in their site-specific relative sea-level rise rates, there is also high uncertainty in the estimates of the cumulative change in site specific sea-level (Table 4.1) using methods 2 and 3. When the mean annual sea-levels were used (Figs. 4.16 and 4.21), relatively short-term variability from El Nino phases is apparent, referring to Fig. 4.1 to identify the dates of the El Nino phases. Despite this variability, generally the same conclusions result from use of full tide gauge time series. These El Nino phase signals are not apparent in the plots showing changes in position of the seaward margins (Fig. 4.15), suggesting that there was no discernible effect of this relatively short term variability on movement of the mangroves' seaward margins.

Based on these observations, and the understanding of the time periods for mangrove species to respond to changes in relative sea-level, use of the full tide gauge 55.6-year data series was likely the most appropriate of the three methods employed. Mangroves likely change position in response to changes in relative sea-level gradually (this study documents mangrove seaward margin migration at a rate of between 0.2 to 0.7 m per decade). Mangrove species gradually colonize new habitat over periods of years and longer via seedling recruitment and vegetative reproduction as new habitat becomes available landward through erosion, inundation, and concomitant change in salinity (Semeniuk, 1994; Duke et al., 1998; Lovelock and Ellison, 2007). Mangrove tree dieback occurs over a similar time frame due to stresses caused by a rising sea-level such as pulses of erosion resulting in weakened root structures and falling of trees, increased salinity, and too high a hydroperiod (Naidoo, 1983; Ellison, 1993, 2000, 2006; Lewis, 2005) (Section 1.5.2). Retreat of the seaward margin of mangroves resulting from a long-term rise in relative sea-level will likely occur through a long-term mean trend of landward transgression, as observed at Masefau and Nu'uuli (Fig. 4.15), rather than change in margin position in response to short-term variability in relative sea-level (SCOR Working Group, 1991; Semeniuk, 1994; List et al., 1997; Pethick, 2001; Pilkey and Cooper, 2004). These short-term episodic events in sea-level range from events that occur over

days such as storms, to seasonal events such as variations in the strengths and prevailing directions of coastal currents and winds, to events that last over a few months such as El Nino and La Nina phases. These factors affect sediment-budget balances, result in short-term changes in sea-level, and result in pulses in erosion and accretion along the seaward margin of mangroves. In summary, over the generally relatively long time periods employed in this study component, of years to decades between remotely sensed images, these observations of long-term mangrove migration when caused by changing relative sea-level are influenced more by the long-term trend in relative sea-level than shorter-term, episodic, cyclical, and small-scale events.

#### **5.7. MANGROVE RESISTANCE AND RESILIENCE TO PROJECTED RELATIVE SEA-LEVEL RISE**

We have now observed changes in mangrove seaward margin positions, identified trends in mangrove sediment surface elevation, observed trends in relative sea-level rise, and assessed whether or not change in sea-level has been a primary control on mangrove position. This prepares us for the next step, to discuss predictions of whether or not the mangroves will keep pace with projected sea-level rise, and for sites where sea-level is predicted to rise relative to the sediment surface, predictions of how the mangroves will respond to rising sea-level. This assesses one aspect of mangrove resistance to sea-level rise (Section 1.1) by determining a mangrove site's predicted ability to keep pace with rising sea-level and maintain vs. alter its current position, with implications for whether or not the ecosystem will experience alteration to its functions, processes and structure (Odum, 1989; Bennett et al., 2005). Similarly, this assesses one aspect of mangrove resilience to sea-level rise, by predicting a mangrove site's capacity to migrate landward in response to rising sea-level and predict the site's future position, with implications for the ecosystem's ability to absorb and reorganize to maintain its functions, processes and structure (Carpenter et al., 2001; Nystrom and Folke, 2001).



If regional relative sea-level continues at the rate observed over the past 55 years of  $1.97 \text{ mm a}^{-1}$ , Nu'uuli mangrove will very likely not keep pace, and continue to migrate landward, i.e., Nu'uuli's threshold is likely already being exceeded and will continue to do so (Section 4.7, Fig. 4.22). The range of site specific relative sea-level rise projections for Nu'uuli are between 152 mm and 885 mm between 2004 and 2100 (Section 4.7, Fig. 4.22). Maesfau mangrove's threshold has been exceeded and is not keeping pace, where, over the past four decades, the seaward margin was observed to migrate landward, and the site will continue to not keep pace over coming decades. There was no linear temporal trend ( $0 \text{ mm a}^{-1}$  change in sediment surface elevation) at Masefau (Fig. 4.17b), indicating a site specific relative sea-level rise rate equivalent to the regional rate as measured by the Pago Pago tide gauge. The range of site specific relative sea-level rise projections for Masefau are between 64 mm and 831 mm between 2004 and 2100 (Section 4.7, Fig. 4.2).

If there is no acceleration in regional relative sea-level rise over the next century (2004-2100), and the trends in sediment surface elevation adopted here are valid, then there will be a net 11.7% reduction in area of the three mangrove study sites. This increases to 47.3% when applying the IPCC upper projection (Section 4.8). The loss of mangrove ecosystem services (Section 1.2), as well as the landward encroachment into developed coastal areas, could be substantial.

Mangrove ecosystems were able to persist through the Quaternary, despite substantial disruptions from large sea-level fluctuations, demonstrating mangroves are highly resilient to change over historic time scales (Woodroffe, 1992). However, over human time scales, mangrove ecosystem's responses to relative sea-level rise over coming decades will not only be a result of the site-specific factor of how sea level is changing relative to the mangrove sediment surface, but resistance and resilience to sea-level rise will also be highly influenced by anthropogenic disturbances. Future mangrove position and area, as well as structure and health, will be affected by several natural processes and anthropogenic stressors other than change in site specific relative sea-level. The Introduction (Sections 1.3 and 1.5.3) identifies myriad climate and non-climate factors that could alter

mangrove position. The occurrence and magnitude of these factors are difficult to predict and potential effects on mangroves, in many cases, are not well understood. For example, human responses to relative sea-level rise and other climate change outcomes have the potential to exacerbate the adverse effects of climate change on mangrove ecosystems. People may construct additional shoreline erosion control structures in response to rising sea level, causing the erosion of mangroves fronting the structure (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997). Or people may increase groundwater extraction in response to increased temperatures and reduced precipitation, resulting in higher mangrove site specific relative sea-level rise (Krauss et al., 2003; Whelan et al., 2005). Non-climate-related factors (both from natural processes and anthropogenic activities) that may cause mass mangrove mortality include excessive sedimentation (Lugo and Cintron, 1975; Hutchings and Saenger, 1987; Ellison, In Press), hydrological blockage causing either sustained or restricted inundation (Hatton and Couto, 1992), sediment erosion, oil spills (Lewis, 1983; Duke et al. 1997), and clearing (Diop, 2003).

The understanding of the synergistic effects of multiple climate change stressors and other anthropogenic and natural stressors on mangroves is also poor. As a result, predictions of year 2100 mangrove positions should be considered rough estimates. Predicted responses of Nu'uuli and Masefau mangroves, which are based on the assumption that relative sea-level rise will continue over the coming century to be a primary control on mangrove position, as determined to be the case over past decades, may prove to be inaccurate if other factors become predominant controls of mangrove position. To assess this, long-term monitoring will enable researchers to establish baselines and distinguish global from local factors inferred to be causing observed changes in mangrove ecosystems (Section 7.2.5).

## **5.8. REGIONAL EXTRAPOLATION**

Based on the observations and predicted mangrove vulnerability and responses to projected relative sea-level rise in American Samoa, an extrapolation for the change in mangrove area through the year 2100 for the

Pacific Islands region were made. Based on relative sea-level trends continuing linearly through the year 2100 with no acceleration, Pacific Island mangroves could experience a reduction in area of 2.4%. The estimate increases to 22.4% when employing IPCC's upper projection for global sea-level rise (about a 0.23% reduction in area per year). Because there is little available quantitative information on trends in area of Pacific Island mangroves due to limited monitoring (Sections 2.1.1 and 6.3.6), we cannot validate the results from this extrapolation from historical observations.

Mangrove vulnerability and responses to changes in relative sea-level are site-specific, as there is variability in rates of change in elevation of mangrove sediment surfaces, and currently, no reliable elevation models for mangrove ecosystems. As a result, this regional extrapolation is an approximate, first-order estimate, and there is high uncertainty in the results.

These regional estimates are weighted heavily by predictions for changes in the four countries containing the majority of the region's mangrove area (Papua New Guinea, Solomon Islands, Fiji and New Caledonia). For instance, under the IPCC upper projection scenario, the estimated reduction in Papua New Guinea mangrove area accounts for 69% of the total regional loss (about 80,000 ha of a total 116,000 ha projected to be lost). However, when looking at the predictions on an individual country/territory basis, under the first scenario, of no acceleration in change in sea-level, 9 of the 16 countries and territories would experience no loss in mangrove area (where it is possible that there could be net gains in mangrove area). Under the upper projection, only two of the countries and territories would not experience losses (Samoa and the Solomon Islands, where large rates of relative sea-level lowering has been observed, Table 4.4).

There is little available quantitative information on trends in area of Pacific Island mangroves due to limited monitoring (Section 6.3.6). However, this study in American Samoa and other observations of mangrove vulnerability to change in relative sea-level, primarily from the western Pacific and Wider Caribbean regions, document that the majority of mangrove sites have not been keeping pace with current rates of relative sea-level rise (Cahoon et al., 2006; Cahoon and Hensel, 2006; McKee et al., 2007).

Longer term studies (e.g., > 10 years) are needed to determine if these are long-term trends vs. cyclical, short-term patterns, and from additional regions to determine if this is a global vs. regional phenomenon.

The global average annual rate of mangrove loss is 1 to 2%, with losses during the last quarter century ranging between 35 and 86% (Valiela et al., 2001; FAO, 2003; Wells et al., 2006; Duke et al., 2007). The estimate of the annual rate of mangrove losses in the Pacific Islands region is 0.9% from 1980-2000, for areas where mangroves are indigenous, slightly below the lower end of the global estimate (FAO, 2006). While, to date, relative sea-level rise has likely been a smaller threat than anthropogenic activities including conversion for aquaculture and filling (IUCN, 1989; Primavera, 1997; Ramsar Secretariat, 1999; Smith et al., 2001; Valiela et al., 2001; Alongi, 2002), results from extrapolating regionally from the American Samoa study suggest that relative sea level rise will be a substantial cause of future reductions in regional mangrove area, causing as high as 25% (0.224% annual losses from rising sea-level [Section 4.9] out of a total of 0.9% total annual losses) of estimated regional losses and as high as 11 to 22% of total estimated annual global losses.

Relative to other Pacific Island countries and territories, American Samoa possesses abundant technical resources that enabled this comprehensive assessment to predict mangrove response to relative sea-level rise (Table 6.1). This presents an opportunity to augment the region's capacity to conduct vulnerability assessments and adaptation options for mangrove response to sea-level rise: Regional capacity can be improved through regional dissemination of the lessons learnt from this study as well as by transferring technical skills and enabling shared resources.

## **Chapter 6**

# **Capacity of Pacific Island Countries and Territories to Assess Vulnerability and Adapt to Mangrove Responses to Climate Change**

### **6.1. INTRODUCTION**

Accurate predictions of changes to coastal ecosystem area and health, including in response to climate change outcomes,, enable the identification of adaptation measures that are appropriate for specific sections of coastline to minimize and offset anticipated losses, and reduce threats to coastal development and human safety (Titus, 1991; Mullane and Suzuki, 1997; Scheffer et al., 2001; Turner, II et al., 2003; Topkins and Adger, 2004; Julius and West, 2008). Island mangroves could experience serious problems due to rising sea level. Over recent decades, the ten Countries and territories in the Pacific Islands region with native mangroves that are experiencing a rise in relative sea level experienced an average rise of 2.0 mm a<sup>-1</sup> (Table 4.4). As observed from this study in American Samoa and previous vulnerability assessments, elevation trends for the majority of mangrove sites are not keeping pace with current rates of regional relative sea-level rise (Table 1.2) (Krauss et al., 2003; Cahoon and Lynch, 1997; Rogers et al., 2005a,b; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2007).

Shoreline development and coastal ecosystems in the Pacific islands region are particularly vulnerable to small increases in sea level and other climate change outcomes. Many of the low islands do not exceed 4 m above current mean sea level, and even on high islands, most development is located on narrow coastal plains. Small island states have limited capacity to adapt to relative sea level rise, including accommodating landward migration of mangroves and other coastal ecosystems. This is a result of their small land mass, high population densities and population growth rates, limited funds, poorly developed infrastructure, and susceptibility to damage from natural disasters (Nurse et al., 2001).

Land-use planners can obtain information from assessments predicting shoreline responses to projected relative sea level rise and other climate change effects over coming decades to mitigate anticipated losses of coastal habitats and avoid and minimize damage to coastal development. An assessment was conducted of the capacity of Pacific Island Countries and territories to determine mangrove vulnerability and predict responses to climate change, and to adapt to mangrove responses to climate change. Results highlight priority national and regional technical and institutional capacity-building needs.

## **6.2. NATIONAL, REGIONAL AND INTERNATIONAL INITIATIVES**

Due to anticipated effects of climate change in the Pacific islands region, several international and regional initiatives discuss the severity of this threat and provide general guidelines for planning. However, there have been few site-based vulnerability assessments or identification of alternatives for site-based management of the responses of mangroves and other coastal ecosystems to sea level rise and other climate change outcomes.

There have been several national or island-scale vulnerability assessments in Pacific Island Countries and territories that provide qualitative assessments to describe the anticipated responses of coastal systems to projected sea level change and other climate change effects and concomitant threats to developed portions of the coastal zone. For example, Phillips (2000) describes the vulnerability of Vanuatu's coastal villages to sea level rise, referencing global sea level rise models produced by the Intergovernmental Panel on Climate Change and other projected global climate change parameters (temperature and precipitation), as a basis for hypothesizing possible environmental, social and economic effects. Other vulnerability assessments have established the locations and elevations of coastal habitats and development and employ global or relative sea level change projections to make rough predictions of the sections of coastline that will be affected. For example, Solomon et al. (1997) created a GIS of a developed stretch of the coastline of Viti Levu, Fiji using topographic maps to identify the locations and elevations of coastal development, including sea

walls, revetments, and other shoreline protection structures. They then used this GIS to assess the vulnerability of the coastline to inundation from four sea level rise scenarios (a rise of 0 m, 0.25 m, 0.5 m, and 1 m) and storm surges. This vulnerability assessment methodology does not account for natural coastal ecosystem responses to changes in relative sea level and other climate change effects.

All twelve Pacific Island Countries with indigenous mangroves are Parties to United Nations Framework Convention on Climate Change (UNFCCC). Of these, eleven countries (Kiribati, Marshall Islands, Federated States of Micronesia, Nauru, Palau, Papua New Guinea, Samoa, Solomon Islands, Tonga, Tuvalu, and Vanuatu) have submitted an initial National Communication to the UNFCCC. Fiji has not yet submitted a National Communication. The eleven initial National Communications discuss general considerations and guidelines related to coastal ecosystem vulnerability and adaptation to future sea level and climate change (Federated States of Micronesia, 1997; Government of Samoa, 1999; Government of Tuvalu, 1999; Kiribati Government, 1999; Republic of Nauru, 1999; Republic of Vanuatu, 1999; Papua New Guinea Government, 2000; Republic of the Marshall Islands Environmental Protection Authority, 2000; Republic of Palau Office of Environmental Response and Coordination, 2002; Solomon Islands Government, 2004; Kingdom of Tonga, 2005). A review of these reports highlights that there is a gap in information on anticipated site-specific responses of mangroves and other coastal ecosystems to climate change effects and site-specific strategies for adaptation. Examples of the general adaptation strategies to manage mangrove and other coastal ecosystem responses to climate change effects identified in National Communication reports include (Federated States of Micronesia, 1997; Government of Samoa, 1999; Republic of Vanuatu, 1999; Kiribati Government, 1999; Papua New Guinea Government, 2000; Republic of the Marshall Islands Environmental Protection Authority, 2000; Republic of Palau Office of Environmental Response and Coordination, 2002; Solomon Islands Government, 2004):

- (i) Establishing zoning rules for setbacks of new development from mangroves;

- (ii) Retreating to higher ground or off-island for appropriate sections of coastline as a last resort option;
- (iii) Identifying sections of coastal areas vulnerable to flooding and inundation to guide future development; and
- (iv) Fortifying relevant sections of developed coastline.

These examples are representative of the broad, general level of specificity in the National Communication reports to assess the vulnerability of and adapt to coastal ecosystem responses to climate change.

The "South Pacific Sea Level and Climate Monitoring Project" was initiated in 1991 to establish stations in eleven Pacific Island Countries to measure the relative motions of land and sea at each station (South Pacific Sea Level and Climate Monitoring Project, 2001). The project is managed by the Oceanography Division of the Australia Bureau of Meteorology (the project was formerly managed by the National Tidal Facility of Flinders University of South Australia). These data will assist in long-term calibration of satellite altimetry and radio astronomy and provide a measure of regional vertical control.

The Secretariat of the Pacific Regional Environment Programme's (SPREP's) Pacific Islands Climate Change Assistance Programme was implemented from 1997-2000 to assist ten Pacific Island Countries, which signed and ratified UNFCCC, with their reporting, training and capacity-building responsibilities under the convention, including assessing their vulnerability to climate change. Participants from 12 countries (Papua New Guinea was the one participating country with indigenous mangroves) received training on assessing climate change vulnerability and adaptation requirements during a six-month training course in 1998. Under this program, SPREP produced a document, *Adapting to Climate Change: Incorporating Climate Change Adaptation into Development Activities in Pacific Island Countries: A Set of Guidelines for Policymakers and Development Planners* (South Pacific Regional Environment Programme, 2000). The document presents general guidelines for Pacific island governments to incorporate considerations of sea level and climate change into new development planning.



The *Regional Wetlands Action Plan for the Pacific Islands* (South Pacific Regional Environment Programme, 1999a) specifies regional actions to monitor mangroves. Action 3.3.1 calls for the development of a regional monitoring program to assess the status of mangroves in the region, evaluate the success of management and conservation actions and develop more effective management practices. Furthermore, Action 3.3.5 identifies that mangroves, particularly those of low islands, are likely to be sensitive to rise in sea level. It promotes the development of a mangrove monitoring network for identification of changes, which has yet to be established.

SPREP implemented the project, "Capacity Building for the Development of Adaptation Measures in Pacific Island Countries," in the Cook Islands, Fiji, Samoa, and Vanuatu from 2002-2005. The aim was to build capacity of communities of these four countries to adapt to climate change, including incorporation of climate change adaptation considerations into national and local planning and budgeting. The project focused on socioeconomic effects and policy development, and did not address coastal ecosystem response to outcomes of climate change, including rising sea level. For instance, SPREP conducted seminars for senior government officials from the four participating countries and produced and distributed briefing papers and educational materials to raise awareness of climate change effects and adaptation (Secretariat of the Pacific Regional Environment Programme, 2003).

The South Pacific Applied Geoscience Commission (SOPAC) has developed an environmental vulnerability index for application at national scales to provide a quick and inexpensive method to characterize the vulnerability of natural systems at large scales. It is not designed for site-based vulnerability assessments (South Pacific Applied Geoscience Commission, 2003).

### **6.3. CAPACITY-BUILDING PRIORITIES TO ADDRESS MANGROVE RESPONSES TO CLIMATE CHANGE**

Information was collected from 10 of the 16 Pacific Island Countries and territories with indigenous mangroves to identify capacity-building priorities to

address mangrove responses to climate change effects (Fig. 6.1): American Samoa - USA, Republic of the Fiji Islands, Republic of Kiribati, Republic of the Marshall Islands, Federated States of Micronesia, Commonwealth of the Northern Mariana Islands - USA, Republic of Palau, Independent State of Papua New Guinea, Kingdom of Tonga, and Republic of Vanuatu. Appendix 3 contains the questionnaire form used to guide the collection of information from each country and territory. These ten countries and territories contain 84% of the area of the region's indigenous mangroves (Table 4.4). Table 6.1 synthesizes the collected information. This section describes the technical and institutional resources requiring strengthening in the Pacific Islands Region, based on the resources needed to assess mangrove vulnerability and responses to climate change (Sections 1.5.6 and 1.5.7, and methods employed in American Samoa in this study, Chapter 3), and capacity to implement alternative adaptation options (Chapter 7).

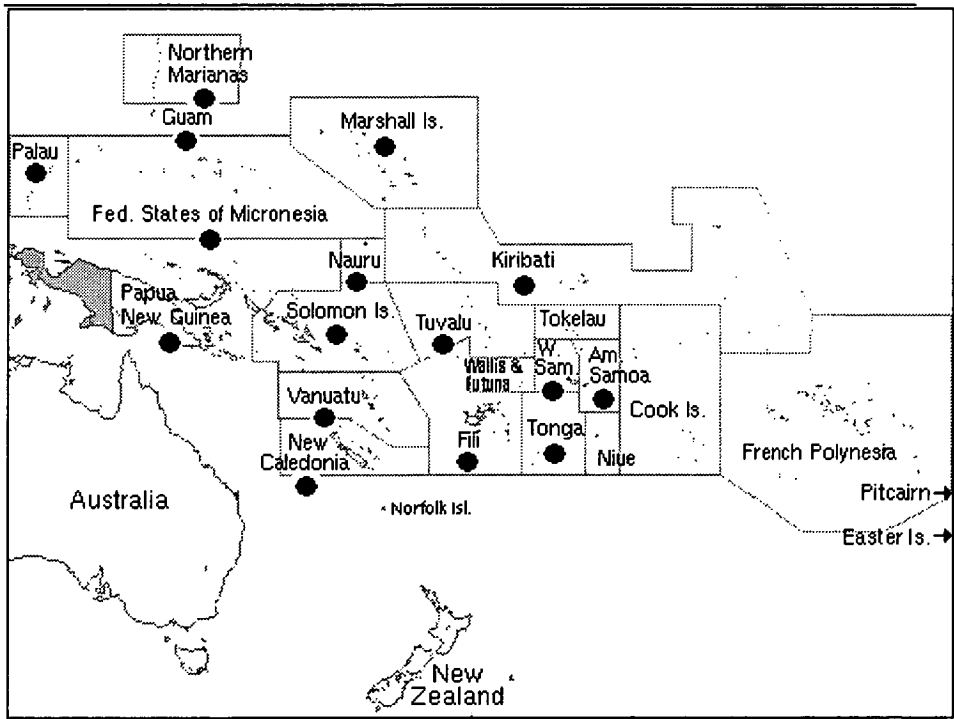


Fig. 6.1. The Pacific Islands region and location of the twelve Pacific Island Countries and four territories with indigenous mangroves, identified by a black oval below the country/territory name.

Table 6.1. Summary of technical and institutional capacity to assess vulnerability and adapt to mangrove responses to relative sea level and climate change, for ten Pacific Island Countries and territories with indigenous mangroves.

Technical and Institutional Capacity Attribute	Country or Territory									
	American Samoa	Fiji	Kiribati	Marshall Islands	Federated States of Micronesia	Northern Mariana Islands	Palau	Papua New Guinea	Tonga	Vanuatu
Length of tide gauge record through Dec. 2005 (years) <sup>a</sup>	56.2	31.2	11.0	58.5	47.7	23.1	33.8	39.7	13.4	10.9
Largest distance between tide gauge and mangrove (km)	6.5	800	Not known	Not known	563	10	700	500	17	Not known
Percent of mangrove boundaries delineated and mapped	100	80	22	0	21	100	99	Not known	90	0
Year of most current mangrove boundary survey <sup>b</sup>	2002	1993	1998	NA	2002	1989	1971	1993	2000	NA
Percent of mangrove islands with topographic map coverage	100	100	22	0	100	100	87.5	86	0	100
Percent of mangrove islands with maps showing buildings, roads, and other development	100	100	25	20	100	100	0	0	33	100
Year of most current map showing location of development <sup>b</sup>	2001	1993	1998	2004	2004	1999	NA	NA	1998	Not known
Date of earliest imagery showing mangrove margin (aerial photos, satellite imagery, maps) <sup>b</sup>	1961	1954	1960	2004	1944	1940	1946	1971	1995	1980
Have mangrove sediment erosion/accretion rates or elevation change rates been measured?	Y	N	N	N	Y	N	N	N	N	N

Is there a mangrove monitoring program	Y	Y	N	N	Y	N	N	N	Y	N
Is there in-country staff with skills to conduct mangrove surveys and inventories?	Y	Y	Y	Y	Y	Y	Y	Y	Y	N
Have mangroves been successfully rehabilitated? <sup>c</sup>	Y, N <sup>d</sup>	Not known	Y	N	N	Y	Y	N	Y	N
Is there a permit or zoning program for coastal development?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Have there been site-specific mangrove vulnerability assessments?	Y	N	N	N	N	N	N	N	N	N
Is there a plan for adaptation to coastal ecosystem responses to climate change?	N	N	N	N	N	N	N	N	N	N
GDP (millions of USD) <sup>e</sup>	12.5 million	2,810	76	144	232	12.5 million	145	4,731	244	341

<sup>a</sup> Tide gauge record lengths based on the time span from the earliest and most current sea level records through December 2005 possessed by the Permanent Service for Mean Sea Level and University of Hawaii Sea Level Center Joint Archive for Sea Level and GLOSS/CLIVAR Research Quality Data Set databases. Does not account for gaps in data records. Tide gauge record lengths may be longer than reported for sites where tide gauges are active, as there is a lag in updating the two data repositories.

<sup>b</sup> Mangrove boundary surveys, maps showing roads and development, and historical imagery may have incomplete coverage of mangroves.

<sup>c</sup> Successful mangrove rehabilitation is defined as one where  $\geq 25\%$  of the project area was successfully restored, enhanced or created.

<sup>d</sup> One effort was successful (38% survival, Appendix 1), a previous effort at the same location was not.

<sup>e</sup> Gross Domestic Product (GDP), the value of all goods and services produced within a nation in a given year, from World Bank data for the year 2005. GDP for the two U.S. territories is for the entire U.S.

Hawaii and Tahiti, where mangroves are human introductions (Allen 1998; Ellison, 1999), are not included in the assessment because management authorities in these areas may actively control the alien invasive species (e.g., Smith, 2005). A Niue government focal point reported that there are no mangrove wetlands in Niue (personal communication, 10 June 2005, Fiafia Rex, Fisheries Division, Niue Department of Agriculture

Forestry & Fisheries). While one true mangrove species *Excoecaria agallocha* is documented to be present in Niue (Yuncker, 1943; Ellison, 1999), in Niue, this species is only found in dry littoral forest.

### **6.3.1. Analysis of Tide Gauge Data for Trends in Mean Sea Level and Extreme High Water Events**

A minimum of a 20-year local tide gauge record is required to obtain an accurate trend in relative sea level (Church et al., 2004a). Many Pacific islands experience tectonic movements that result in substantial differences in local sea level relative to global eustatic trends. In addition to vertical land-level changes from tectonics, change in relative mean sea level over time as measured by a tide gauge can result from subsidence from extraction of subsurface groundwater or oil, oceanographic processes such as El Niño phases and changes in offshore currents, long-term changes in regional temperature, sediment consolidation, as well as from global sea level change (Komar, 1998; Church et al., 2001). The closer the tide gauge is to the mangrove site, the more accurately it will reflect the actual sea level changes that are affecting the mangroves. For sites with a local tide gauge record of < 20 years or sites that are located too far from a tide gauge, sea level trends can be calculated using the near global coverage of satellite altimetry data combined with historical global tide gauge records (Church et al., 2004a). Relative sea-level as measured by the tide gauge may be substantially different from trends in sea-level at the mangrove sites due to differences in local tectonic processes, coastal subsidence, sediment budgets, and meteorological and oceanographic factors. To account for this possibility, to calculate any difference in trends in relative sea-level between the tide gauge and mangrove sites, local benchmarks should be established adjacent to each mangrove study site, and surveying should occur from the local benchmarks to a level network benchmark, which is related to a mean sea-level datum, using geodetic leveling techniques (American Society of Civil Engineers, 2000). Pacific Island Countries and territories also need the technical capacity to analyze available tide gauge data to determine projected trends in mean sea level and extreme high water events, and incorporate this information into land-use planning.

Most (7 of 10) countries and territories have sufficiently long tide gauge records ( $\geq 20$  years) to determine accurate trends in mean sea level and extreme high water events (Fig. 6.2). However, tide gauges of four of the seven countries and territories with the long ( $\geq 20$  years) tide gauge records are located several hundred kilometers from at least one mangrove site in that country. Three countries with tide gauge records  $< 20$  years require assistance to determine reconstructed sea level trends from satellite altimetry data combined with historical global tide gauge records. Agencies managing coastal land use and coastal ecosystems of Pacific Island Countries and territories require assistance to determine trends in relative mean sea level and trends in the frequency and elevations of extreme high water events (Church et al., 2001; Woodworth and Blackman, 2002, 2004), and assistance to interpret and incorporate this information into land-use planning processes.

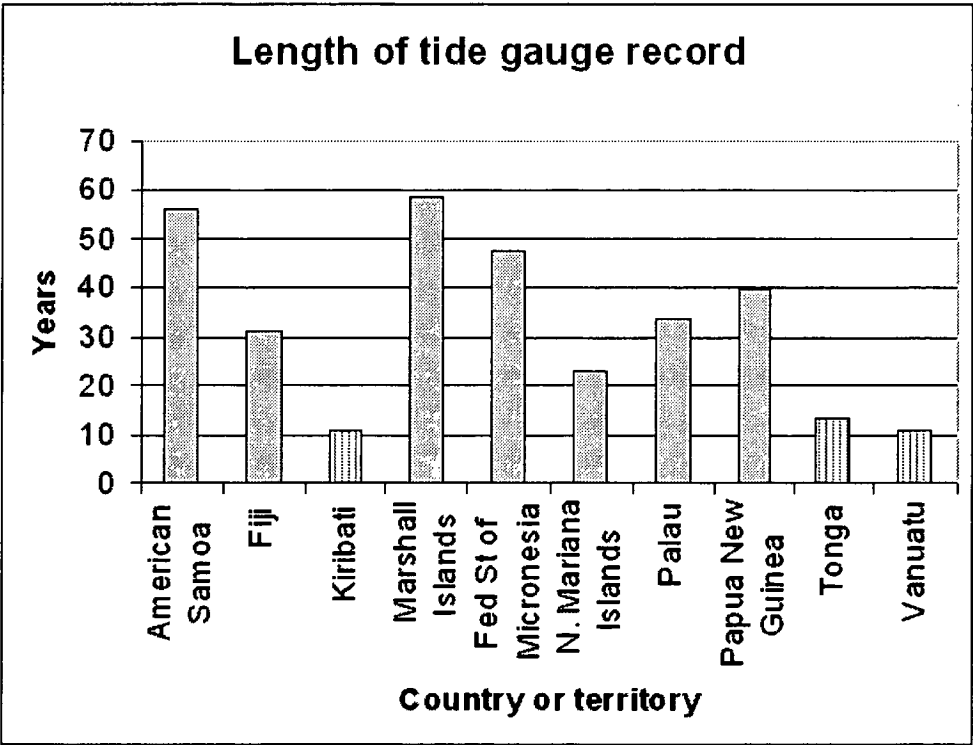


Fig. 6.2. Length of tide gauge records of ten Pacific Island Countries and territories with indigenous mangroves. Bars with vertical red lines have a tide gauge record  $< 20$  years.

### **6.3.2. Information on Change in Sea Level Relative to Mangrove Surfaces**

Information on trends in the change in elevation of the mangrove surface is needed to determine how sea level has been changing in recent decades relative to the mangrove surface. Section 1.5.6 describes alternative methods to obtain this information.

In the Pacific Islands region, besides the assessment conducted here in American Samoa, information on sea level rise rates relative to mangrove sediment surfaces and rates of change in sediment surface elevation are available only for the Federated States of Micronesia (Krauss et al., 2003). Otherwise, regionally there is a lack of information on how sea level is changing relative to mangrove surfaces. Observations of trends in change in the elevation of the mangrove surface are needed over at least several years, but possibly over decades (Sections 5.5.1, 8.3.3).

### **6.3.3. Analysis of Historical Imagery to Observe Changes in Mangrove Margin Positions**

Analysis of a time series of recent historical aerial photographs and satellite imagery showing the positions of mangrove seaward margins can be used to measure any linear temporal trends in horizontal movement of the mangrove margins (erosion or seaward progression). This information can then be used to determine if the movement has been correlated with observed rates of change in relative mean sea level, and predict the future position of the seaward margin. The longer the length of the time series and number of remotely sensed images available, the more accurately a linear trend can be detected.

Eight countries and territories have historical imagery  $\geq 25$  years old (Fig. 6.3). However, analysis to detect a linear trend in horizontal position of seaward mangrove margins has only been conducted for American Samoa in this study. Assistance with acquiring historical aerial photographs and IKONOS and QuickBird satellite imagery, and technical assistance with co-registering available historical imagery to the georeferenced satellite imagery is needed. Some countries and territories may initially require assistance to establish or augment the capacity of GIS programs.

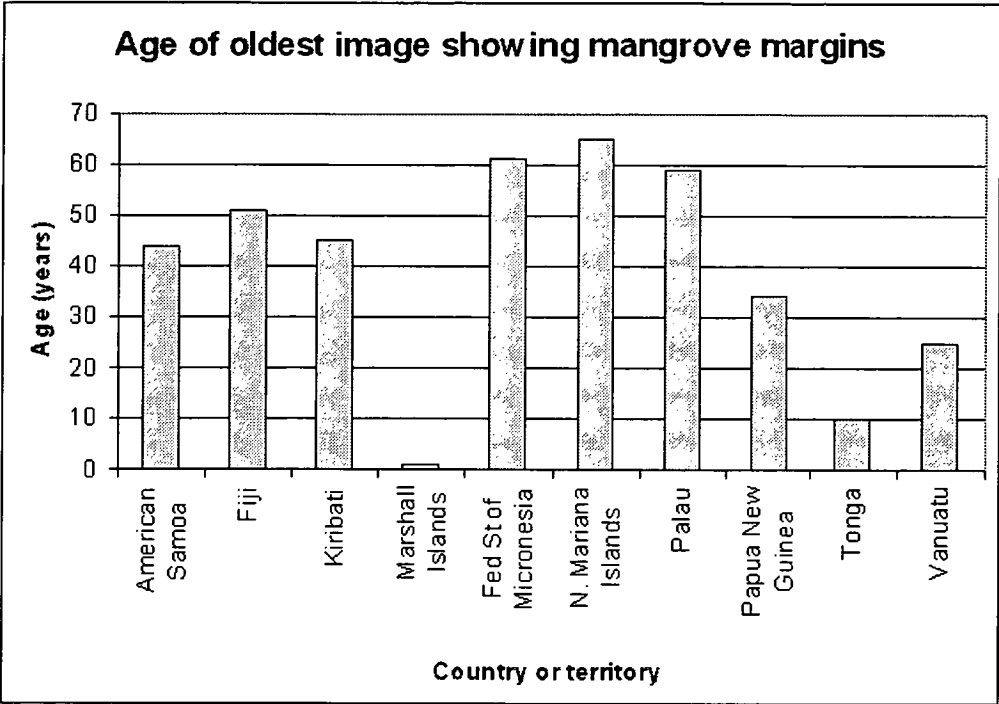


Fig. 6.3. Age of oldest known historical imagery showing the location of mangrove margins, for ten Pacific Island Countries and territories with indigenous mangroves.

6.3.4. Mangrove Boundary Delineation

Periodic delineation of the mangrove landward margin using GPS or traditional survey techniques is needed to observe any movement of the boundary, providing fundamental information needed to monitor trends in mangrove area. Interpretation of remotely sensed imagery (aerial photos and space imaging) generally can be used to delineate the mangrove seaward margin (Section 1.5.7). Otherwise, delineation with GPS or EDM (electronic distance measurement) surveying can be conducted. The *World Atlas of Mangroves*, originally published in 1997 (Spalding et al., 1997) and now being updated (a second edition is due out in late 2007), includes information on the status and trends in mangrove area.

Five of nine countries and territories have delineated 80% or more of their mangrove boundaries (Fig. 6.4). Two countries have not delineated mangrove boundaries. While Papua New Guinea has delineated mangrove boundaries, the percent that has been delineated was not available, and is



not included in Fig. 6.5. Four of the ten countries and territories have delineated mangrove boundaries within the last ten years. This highlights the need for some countries and territories to delineate mangrove boundaries at regular intervals. To prevent a net loss in mangrove ecosystems, resource managers require information obtained from continuous monitoring programs that collect data on mangrove area and health at time intervals that permit sustainable management (Gilman, 1999a). Managers need information from mangrove boundary delineations as well as results of monitoring of functional processes at least every few years in order to understand trends in mangrove quantity and quality. Of the eight countries included in this assessment, the four with the lowest percent coverage of mangrove boundary delineation include the three of the four poorest countries, based on Gross Domestic Product (GDP) (Table 6.1). This suggests that financial constraints might be a main cause of a government's inability to conduct mangrove boundary delineations.

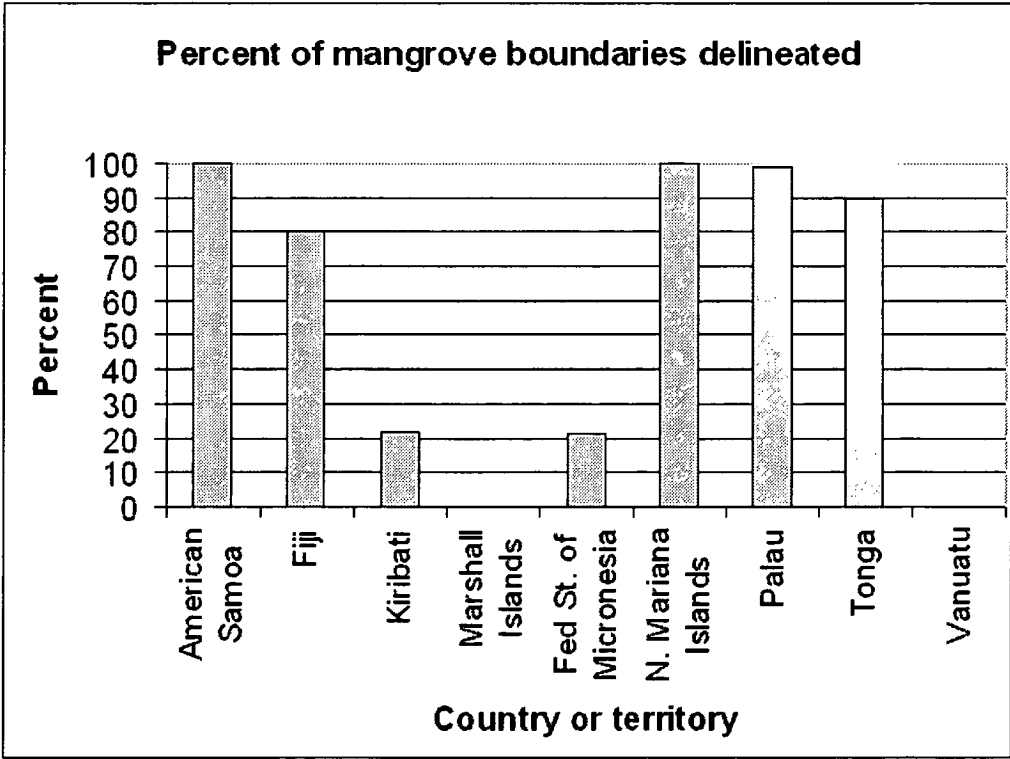


Fig. 6.4. Percent of total mangrove area with boundaries delineated of nine Pacific Island Countries and territories with indigenous mangroves.

### **6.3.5. Map Products**

Topographic information is needed to determine the mean slope of the land immediately adjacent to the landward mangrove margins in order to estimate rates of landward mangrove migration. This requires a recent delineation of the landward mangrove margin. Alternative scenarios for projected change in sea level relative to the mangrove surface can then be used to estimate the distance that the landward mangrove margin will migrate. If sea level is rising relative to the mangrove surface, then information on the current location of any obstacles to the landward migration of mangroves, such as seawalls, buildings, and roads, and the distance that these structures are from the current landward mangrove margin, is needed to determine how these structures may obstruct future landward mangrove migration.

Some countries identified a need for topographic maps and maps showing the location of roads and development in the vicinity of mangroves. Support for in-country GIS programs may be needed to produce these map products. Seven of the countries and territories have > 85% topographic map coverage of their islands containing mangrove wetlands (Fig. 6.5). Two countries lack any topographic map coverage of their islands containing mangroves. Five countries and territories have maps showing locations of development and roads in the vicinity of all mangroves (Fig. 6.6). Two countries lack maps showing development and roads next to any of their mangroves.

Of the nine countries included in this assessment related to map products, the three with consistently low results (Kiribati, the Marshall Islands and Tonga) include the two poorest countries (Table 6.1). This suggests that financial constraints might be a main cause of a government's inability to obtain these map products.

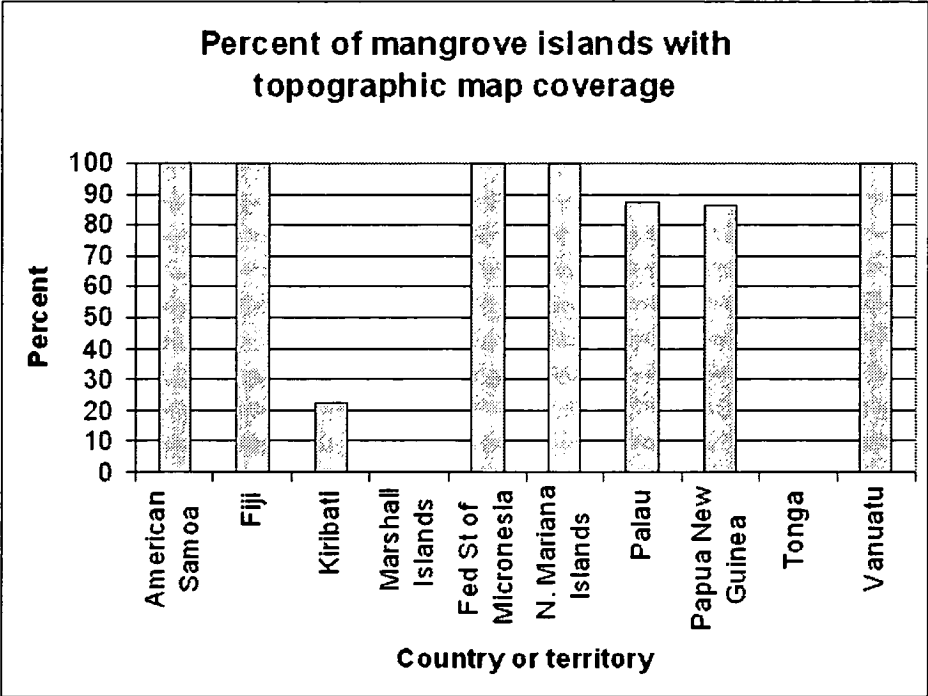


Fig. 6.5. Percent of the islands with mangroves that also have topographic map coverage, for ten Pacific Island Countries and territories with indigenous mangroves.

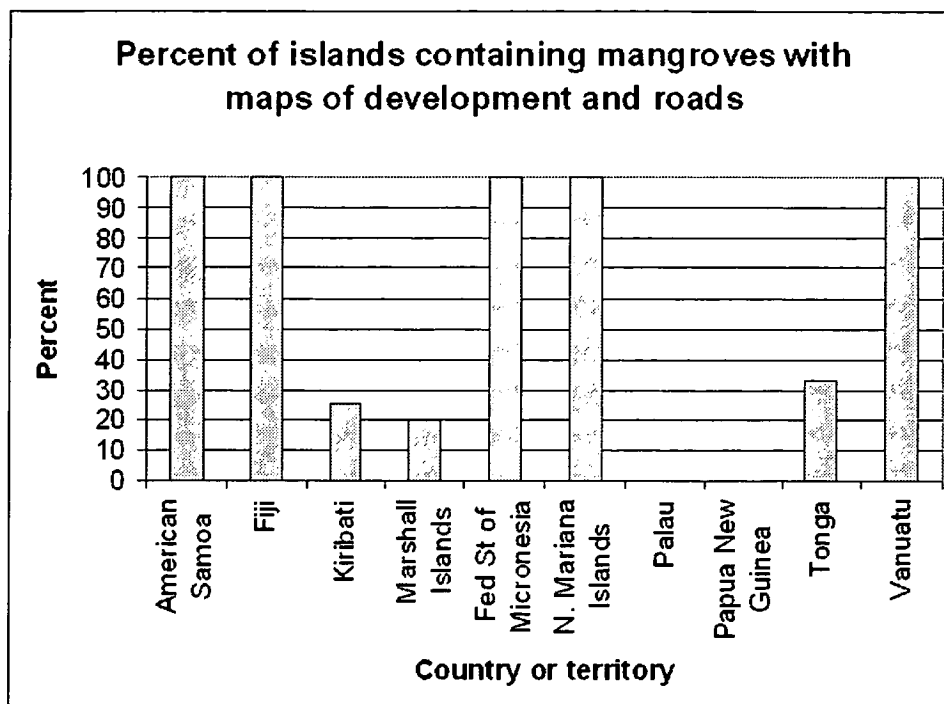


Fig. 6.6. Percent of islands containing mangroves for which maps showing the location of development and roads are available, for ten Pacific Island Countries and territories with indigenous mangroves.

### 6.3.6. Mangrove Monitoring and Assessment for Adaptive Management and Regional Mangrove Monitoring Network

In-country staff with training, experience, and motivation is required to conduct monitoring and assessment of relevant mangrove parameters, in part, to facilitate adaptive management. Sea level and climate changes are expected to alter mangrove position, area, structure, species composition, and health. Linking national mangrove monitoring efforts through a regional network using standardized techniques would enable the separation of site-based influences from global changes to provide a better understanding of mangrove responses to global climate and sea level change and alternatives for mitigating adverse effects (Ellison, 2000; Nurse et al., 2001). Section 7.2.5 identifies parameters and monitoring techniques that should be included in a regional monitoring network.

Four countries and territories identify a strong need for training and capacity-building of in-country personnel in mangrove assessment and monitoring: Vanuatu, Kiribati, Northern Mariana Islands, and Palau. Other

countries (American Samoa, Tonga, Fiji, Marshall Islands, Papua New Guinea, and the Federated States of Micronesia) have some in-country capacity already, but identify information gaps. Fiji, Federated States of Micronesia and American Samoa conduct some monitoring of mangrove tree girth (diameter at breast height or DBH), which allows quantitative assessment of mangrove ecosystem change in community structure and growth rates. Palau and Tonga have more limited monitoring, such as of birds or human impacts.

There has also been no coordination between the limited mangrove monitoring work that has been done. The countries and territories with a mangrove monitoring program do not employ regionally standardized techniques to enable a meaningful comparison of results from the different programs. Other countries have no monitoring. There is no Pacific Islands regional mangrove monitoring program in place. Establishing a regional wetland monitoring network for the Pacific Islands region has been proposed in the *Action Strategy for Nature Conservation in the Pacific Islands Region* (South Pacific Regional Environment Programme, 1999a), and the *Regional Wetlands Action Plan for the Pacific Islands* (South Pacific Regional Environment Programme, 1999b). All countries indicate that they would be interested in participating in a regional network to monitor mangroves and assess mangrove response to sea level rise and climate change, if such a network were established.

Establishing mangrove baselines and monitoring gradual changes through regional networks using standardized techniques will enable the separation of site-based influences from global changes to provide a better understanding of mangrove responses to climate change, and alternatives for mitigating adverse effects (Ellison, 2000; Nurse et al., 2001). The monitoring system, while designed to distinguish climate change effects on mangroves, would also therefore show local effects, providing coastal managers with information to abate these sources of degradation.

#### **6.3.7. Strengthen Management Frameworks**

Governments need the institutional capacity to manage a land-use permit or zoning program to ensure coastal earthmoving and development activities

are sustainable, including accounting for effects on mangroves, and to plan for any landward mangrove migration. While existence of a coastal permit and zoning program does not necessarily mean that current legal and management frameworks and political will are adequately preventing mangrove degradation, it indicates that the institutional capacity needed to sustainably manage activities in mangroves and other sensitive coastal ecosystems exists. Given an existing coastal development permit or zoning program, if the political will exists, it could be possible to establish zoning setbacks for new coastal development adjacent to mangroves in certain sections of coastline, adopt rules on where hard versus soft engineering erosion control structures can and cannot be constructed, and determine which sections of coastline would undergo managed retreat versus fortification.

All ten participating countries and territories report having some form of coastal permitting or zoning program that regulates coastal activities, such as earthmoving and development activities. The existence of a framework to manage coastal activities is part of the requisite institutional capacity to sustainably manage activities in mangroves and other sensitive coastal ecosystems. However, this does not necessarily mean that current legal and management frameworks and political will are adequately preventing mangrove degradation. There is a need to assess the efficacy of national management frameworks at preventing mangrove degradation to determine if this is an area in need of attention. For instance, despite the existence of a permit program for coastal development activities, the wetlands management framework in the U.S. Commonwealth of the Northern Mariana Islands has not been preventing site-specific, island-wide, or cumulative losses of wetland functional performance or wetland area (Gilman, 1998, 1999b). Also, for example, Palau's state Public Land Authorities have been leasing and allowing the development of property containing mangroves (Republic of Palau Office of Environmental Response and Coordination, 2002).

Only American Samoa, through this study, has assessed the site-specific vulnerability of mangroves to sea level. Information from the case studies and a review of National Communication Reports to the United Nations Framework Convention on Climate Change reveal that none of the

ten countries and territories have developed a plan for adaptation to mangrove or other coastal ecosystem responses to climate change effects. Technical assistance is needed to support conducting site-specific vulnerability assessments and to incorporate resulting information into land-use and master planning.

#### **6.3.8. Mangrove Rehabilitation**

Capacity to rehabilitate (restore, enhance and create) mangroves will complement adaptation to mangrove response to sea level and climate change. Restoring areas where mangrove habitat previously existed, enhancing degraded mangroves by removing stresses that caused their decline, and creating new mangrove habitat will help to offset anticipated reductions in mangrove area and increase resilience to climate change effects. If successful mangrove rehabilitation has been achieved in the past, this indicates that it may be possible to replicate this success at other sites. However, failure to provide adequate training to coastal managers in the basics of successful mangrove rehabilitation leads to project failures or projects that only partially achieve stated goals (Lewis, 2005).

There has been limited activity in the region in rehabilitation of mangroves, with small-scale successful projects only recorded from Kiribati, Northern Mariana Islands, Palau, Tonga, and American Samoa (Appendix 1), and two failed mangrove rehabilitation efforts in American Samoa and Papua New Guinea. The results of two additional rehabilitation efforts, in Palau and Fiji, are not known. This highlights the need for improved staff training, capacity building and information sharing.

#### **6.4. FINANCING**

The cost to implement the methods employed in American Samoa to assess mangrove vulnerability and responses to site-specific trends in relative sea-level (Sections 1.5.6, 1.5.7, 3.1, 3.3, 3.4, 3.6-3.9), for equipment, labor (paid and volunteer), office expenses and overhead over a two-year period is estimated to have been USD \$63,500. Labor comprised 79% of the total estimated costs. Of this total, direct costs to implement the project in

American Samoa over the two-years was \$23,500: The three American Samoa government staff who contributed to the project did not require additional financing to participate, as these staff are permanent civil service employees whose salaries are covered through annual government appropriations. Replicating these methods in less developed countries and territories of the Pacific Island region would cost less due to lower labor costs than in American Samoa. It would be beneficial to conduct the monitoring of elevation stakes in perpetuity, with monitoring occurring about four times annually, and data analysis being updated biennially. It would be beneficial to repeat the assessment periodically, perhaps every five years, to make use of new satellite imagery, longer tide gauge data series, human modifications to adjacent land uses, and new projections for global sea-level rise trends.

A mixture of public and private financing, government funding, and income-generation by management authorities are potential sources of long-term funding (IUCN and European Commission, 1999). In most countries in the region, external financial support would likely be needed, at least for initial years of program establishment and implementation (Nickerson-Tietze, 2001). Potential external funding sources include United Nations Global Environment Facility perhaps managed by the Secretariat of the Pacific Regional Environment Programme, bi-lateral development cooperation agencies, private foundations (e.g., Packard, Pew), Secretariat of the Pacific Community, private companies (e.g., International Tropical Timber Organization), and international non-governmental organizations (e.g., Conservation International, WWF-International, IUCN) (Phillips, 2000b). Alternative national-level funding mechanisms include taxes, levies, surcharges, and tax incentives; tax deduction schemes; grants from private foundations; national environmental funds; debt swaps; national and provincial lotteries; public-good service payments; and workplace donation schemes (Phillips, 2000b). Site-level funding mechanisms include user fees, cause-related marketing, adoption programs, corporate donations, individual donations, planned giving, and site memberships (Phillips, 2000b).



## **Chapter 7**

### **Adaptation Options for Mangrove Responses to Climate Change**

#### **7.1. INTRODUCTION**

Adaptation activities can be taken to attempt to increase the resistance and resilience of mangroves to climate change (Scheffer et al., 2001; Turner, II et al., 2003; Topkins and Adger, 2004; Julius and West, 2008). Mangrove ecosystems were able to persist through the Quaternary despite substantial disruptions from large sea-level fluctuations, demonstrating that mangroves are highly resilient to change over historic time scales (Woodroffe, 1987, 1992). However, over coming decades, mangrove vulnerability and responses to climate change will be highly influenced by anthropogenic disturbances, including direct sources of degradation such as clearing and filling, and human responses to climate change that adversely affect mangroves (Section 1.3.8). Measures can be taken to avoid and minimize these anthropogenic sources of stress, which reduce mangrove resistance and resilience to rising sea-level.

Management authorities, especially of small island countries and territories, are encouraged to determine predictions of coastal ecosystem responses to projected relative sea-level rise. This will provide the requisite information to enable the identification and adoption of appropriate adaptation measures. This should be conducted before adverse effects of climate change become apparent to human coastal communities, to provide adequate lead time to minimize social disruption and cost, minimize losses of valued coastal habitats, and maximize available options. Our research on American Samoa's major mangroves found landward migration with relative sea level rise over the past four decades, with projections for as high as 47% reductions in area by the year 2100. A 22% reduction in mangrove area is possible in the Pacific Islands region. Effects from increased extreme high water levels and frequency may exacerbate anticipated reductions in mangrove area and health in some parts of the Pacific (Church et al., 2006).

Reduced mangrove area and health and landward mangrove migration will increase the threat to human safety and shoreline development from coastal hazards such as erosion, flooding, and storm waves and surges. Predicted mangrove losses will also reduce coastal water quality, reduce biodiversity, eliminate fish nursery habitat, adversely affect adjacent coastal habitats (Mumby et al., 2004), and eliminate a major resource for human communities that traditionally rely on mangroves for numerous products and services (Satele, 2000; Gilman and Sauni, 2005). Adaptation measures can reduce the risk of adverse outcomes from predicted mangrove responses to climate change (Scheffer et al., 2001; Turner, II et al., 2003; Topkins and Adger, 2004).

## **7.2. ALTERNATIVE ADAPTATION ACTIONS**

Alternative adaptation approaches include (Scheffer et al., 2001; Turner, II et al., 2003; Barber et al., 2004; Topkins and Adger, 2004; Julius and West, 2008):

- Managed retreat or fortification, as determined for specific section of coastline through site-planning;
- “No regrets” reduction of anthropogenic stresses (e.g., managing activities in the catchment that affect mangrove sediment elevation), in part, to augment mangrove resistance and resilience to projected climate change outcomes;
- Protected areas, including through representation, replication, refugia, site selection that accounts for predicted responses to climate change, and systems of networks to protect connectivity between ecosystems;
- Mangrove rehabilitation (Appendix 1), including enhancement to augment resistance and resilience, and ecological restoration to offset anticipated losses;
- Monitoring to detect changes in baseline conditions, to distinguish global from local factors inferred to be causing the observed changes, and to inform decisions for timely adaptation actions; and
- Outreach and education to augment community support for instituting adaptation actions.

Alternative options for adaptation for climate-sensitive ecosystems, including mangroves, are summarized in Table 7.1. Subsequent sections provide more detailed descriptions of the alternative adaptation strategies.

Table 7.1. Adaptation options to augment mangrove resistance and resilience to climate change.

Adaptation Option	Description
"No regrets" reduction of stresses	Eliminate non-climate stresses on mangroves (e.g., filling, conversion for aquaculture, pollution) in order to augment overall ecosystem health, in part, to reduce mangrove vulnerability to and increase resilience to stresses from climate change. These "no regrets" mitigation actions are justified and beneficial even in the absence of adverse effects on mangroves from climate change (Adger et al., 2007; Julius and West, 2007).
Manage activities in catchment that affect mangrove sediment elevation	In order to attempt to augment mangrove resistance to sea-level rise relative to the mangrove sediment surface, activities within the mangrove catchment can be managed to minimize long-term reductions in mangrove sediment elevation, or enhance sediment elevation. For instance, limiting development of impervious surfaces within the mangrove catchment and managing rates and locations of groundwater extraction can reduce alteration to natural groundwater recharge to the mangrove systems, which might be an important control on mangrove elevation. Also, avoiding and limiting human activities that reduce mangrove soil organic matter accumulation, such as the diversion of sediment inputs to mangrove systems, nutrient and pollutant inputs into mangroves, and mangrove timber harvesting can contribute to maintaining relatively natural processes which control trends in sediment elevation. Depending on the tree species and nutrient added, nutrient enrichment can affect mangrove productivity, changing root production and organic material inputs, changing the rate of change in sediment elevation (Feller et al., 2003; McKee et al., 2002, 2007). Enhancement of mangrove sediment accretion rates, such as through the beneficial use of dredge spoils, could augment mangrove sediment elevation (Lewis, 1990), but would need to be implemented carefully so as to avoid excessive or sudden sediment deposition (Ellison, 1998).
Managed retreat	Site planning for some sections of shoreline containing mangroves, such as areas that are not highly developed, may facilitate long-term retreat with relative sea level rise (Dixon and Sherman, 1990; Mullane and Suzuki, 1997). "Managed retreat" involves implementing land-use planning mechanisms before the effects of rising sea level become apparent, which can be planned carefully with sufficient lead time to enable economically viable, socially acceptable and environmentally sound management measures. Coastal development could remain in use until the eroding coastline becomes a safety hazard or begins to prevent landward migration of mangroves, at which time the development can be abandoned or moved inland. Adoption of legal tools, such as rolling easements, can help make eventual abandonment more acceptable (Titus, 1991). Zoning rules for building setbacks and permissible types of new development can be used to reserve zones behind current mangroves for future mangrove habitat. Managers can determine adequate setbacks by assessing site-specific rates for landward migration of the mangrove landward margin. Construction codes can plan for mangrove landward migration based on a desired lifetime for coastal development (Mullane and Suzuki, 1997). Any new construction of minor coastal development structures, such as sidewalks and boardwalks, could be required to be expendable with a lifetime based on the assessed sites' erosion rate and selected setback. Rules could prohibit construction of coastal engineering structures, which obstruct natural inland migration of mangroves. This managed coastal

retreat will allow mangroves to migrate and retain their natural functional processes.

- Fortification** While mangroves provide natural coastal protection that is expensive to replace with artificial structures (Mimura and Nunn, 1998), for some sections of highly developed coastline adjacent to mangroves, site planning may justify use of hard engineering technology (e.g., groins, seawalls, revetments, bulkheads) and other shoreline erosion control measures (e.g., surge breakers, dune fencing, detached breakwaters) to halt erosion. As a result, mangrove ecosystem services will gradually be reduced: The structure will prevent the mangroves' natural landward migration and the mangrove fronting the structure, as well as immediately downstream in the direction of longshore sediment transport from the structure, will eventually be converted to deepwater habitat (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997; Mimura and Nunn, 1998).
- Representation, replication and refugia through a system of protected area networks** Protected areas can be established and managed to implement mangrove representation, replication and refugia. Ensuring representation of all mangrove community types when establishing a network of protected areas and replication of identical communities to spread risk can increase chances for mangrove ecosystems surviving climate change and other stresses (Julius and West, 2007). Ensuring that a portfolio of each different community type is represented is a strategy for optimizing climate change resilience as this representation increases the change that at least one of these communities with disparate physical and biological parameters will survive climate change stressors and provide a source for re-colonizing. Replication, through the protection of multiple areas of each mangrove community type, by protecting multiple examples of each vegetation zone and geomorphic setting can help avoid the loss of a single community type (Roberts et al., 2003; Salm et al., 2006; Wells, 2006). Protected area selection can include mangrove areas that act as climate change refugia, communities that are likely to be more resistant to climate change stresses (Palumbi et al., 1997; Bellwood and Hughes, 2001; Salm et al., 2006). For instance, mature mangrove communities will be more resistant and resilient to stresses, including those from climate change, than recently-established forests. Protecting refugia areas that resist and/or recover quickly from disturbance in general, or that are predicted to be able to keep pace with projected relative sea-level rise can serve as a source of recruits to re-colonize areas that are lost or damaged.
- Protected area site selection should account for predicted ecosystem responses to climate change (Barber et al., 2004). For instance, planners need to account for the likely movements of habitat boundaries and species ranges over time under different sea level and climate change scenarios, as well as consider an areas' resistance and resilience to projected sea level and climate changes and contributions to adaptation strategies. Site-specific analysis of resistance and resilience to climate change when selecting areas to include in new protected areas should include, for example, how discrete coastal habitats might be blocked from natural landward migration, and how severe are threats not related to climate change in affecting the site's health.
- A system of networks of protected areas can be designed to protect connectivity between coastal ecosystems, including mangroves (Crowder et al., 2000; Stewart et al., 2003; Roberts et al., 2001, 2003). Protecting a series of mature, healthy mangrove sites along a coastline could increase the likelihood of there being a source of waterborne seedlings to re-colonize sites that are degraded. Protected area designs should include all coastal ecosystems to maintain functional links (Mumby et al., 2004).

Mangrove rehabilitation	Mangrove enhancement (removing stresses that caused their decline) can augment resistance and resilience to climate change, while mangrove restoration (ecological restoration, restoring areas where mangrove habitat previously existed) (Kusler and Kentula, 1990; Lewis, 2005; Lewis et al., 2006) can offset anticipated losses from climate change.
Regional monitoring network	Given uncertainties about future climate change and responses of mangroves and other coastal ecosystems, there is a need to monitor and study changes systematically. Establishing mangrove baselines and monitoring gradual changes through regional networks using standardized techniques will enable the separation of site-based influences from global changes to provide a better understanding of mangrove responses to sea level and global climate change, and alternatives for mitigating adverse effects (CARICOMP, 1998; Ellison, 2000). For instance, coordinated observations of regional phenomena such as a mass mortality event of mangrove trees, or trend in reduced recruitment levels of mangrove seedlings, might be linked to observations of changes in regional climate, such as reduced precipitation. The monitoring system, while designed to distinguish climate change effects on mangroves, would also therefore show local effects, providing coastal managers with information to abate these sources of degradation (a “no-regrets” adaptation approach).
Outreach and education	Outreach and education activities can augment community support for adaptation actions. The value of wetlands conservation is often underestimated, especially in less developed countries with high population growth and substantial development pressure, where short-term economic gains that result from activities that adversely affect wetlands are often preferred over the less-tangible long-term benefits that accrue from sustainably using wetlands. Education and outreach programs are an investment to bring about changes in behavior and attitudes by having a better informed community of the value of mangroves and other ecosystems. This increase in public knowledge of the importance of mangroves provides the local community with information to make informed decisions about the use of their mangrove resources, and results in grassroots support and increased political will for measures to conserve and sustainably manage mangroves.

### **7.2.1. Site-planning, Managed Retreat vs. Fortification**

The site-specific approach for adaptation to coastal ecosystem responses to relative sea-level rise and other climate change stressors will be made as part a broader coastal planning analysis, where mitigation actions are typically undertaken to address both climate and non-climate threats (Gilman, 2002; Adger et al., 2007). This analysis requires balancing multiple and often conflicting objectives of allowing managers and stakeholders to sustain the provision of ecological, economic, and cultural values; address priority threats to natural ecosystem functioning; maintain ecological processes and biodiversity; achieve sustainable development; and fulfill institutional, policy, and legal needs (Gilman, 2002). Community-based collaborative management approaches, which capitalize on traditional knowledge and management systems, and catalyze stakeholder support for requisite conservation measures, are suitable in some areas of the Pacific Islands region (Gilman, 1997, 2002).

Site planning for some sections of shoreline containing mangroves that are not highly developed may call for gradual, long-term retreat with relative sea level rise (Dixon and Sherman, 1990; Mullane and Suzuki, 1997; Ramsar Bureau, 1998). “Managed retreat” involves implementing land-use planning mechanisms before the effects of rising sea level become apparent, which can be planned carefully with sufficient lead time to enable economically viable, socially acceptable, and environmentally sound management measures. These objectives are consistent with American Samoa’s coastal land use rules, which require a minimum of a 7.6 m (25 foot) buffer from wetlands. With managed retreat, coastal development could remain in use until the eroding coastline becomes a safety hazard or begins to prevent landward migration of mangroves, at which time the development can be abandoned or moved inland. Adoption of legal tools, such as rolling easements, can help make such eventual coastal abandonment more acceptable to coastal communities (Titus, 1991). Zoning rules for building setbacks and permissible types of new development can be used to reserve zones behind current mangroves for future mangrove habitat. Managers can determine adequate setbacks by assessing site-specific rates for landward migration of the mangrove landward margin. Construction codes can be instituted to account for relative sea level rise rate projections to allow for the

natural inland migration of mangroves based on a desired lifetime for the coastal development (Mullane and Suzuki, 1997). Any new construction of minor coastal development structures, such as sidewalks and boardwalks, would be required to be expendable with a lifetime based on the assessed sites' erosion rate and selected setback. Otherwise, the structure should be portable. Rules could prohibit landowners of parcels along these coasts from constructing coastal engineering structures to prevent coastal erosion and the natural inland migration of mangroves. This managed coastal retreat will allow mangroves to migrate and retain their natural functional processes, including protecting the coastline from wind and wave energy.

Employing shoreline erosion control measures, such as surge breakers, dune fencing, and detached breakwaters, can help reduce the rate of coastal erosion (Mullane and Suzuki, 1997). Use of hard engineering technology, including groins, seawalls, revetments, and bulkheads, all traditional responses to coastal erosion and flooding in small island states (e.g., 16.5% - 68.0% of the three American Samoa mangrove study sites' landward margins have adjacent coastal development) and worldwide, can increase coastal vulnerability (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997; Mimura and Nunn, 1998; Nurse et al., 2001). These coastal engineering structures usually can effectively halt erosion as relative sea-level rises, but often lead to the loss of the coastal system located in front of and immediately downstream in the direction of longshore sediment transport from the structure, converting the seaward coastal system into deepwater habitat (Tait and Griggs, 1990; Fletcher et al., 1997; Mullane and Suzuki, 1997). For some sites, it may be less expensive to avoid hard solutions to relative sea level rise and instead allow coastal ecosystems to migrate inland. These ecosystems provide natural coastal protection that may be more expensive to replace with artificial structures (Mimura and Nunn, 1998; Ramsar Bureau, 1998). However, results of site planning may justify use of hard engineering technology and shoreline erosion control measures to prevent erosion for some sections of highly developed coastline adjacent to mangroves. As a result, the mangroves' natural landward migration will be prevented and the mangrove fronting the development will eventually be lost, along with its valued function of buffering the developed coastline from wave and wind energy.



Most cost-benefit analyses included in site planning only examine costs and benefits as measured by market prices, ignoring mangrove and other coastal system values not described by established monetary indicators (Dixon and Sherman, 1990; Ramsar Bureau, 1998). Site planning and cost-benefit analyses employed to determine if a section of coastline abutting a mangrove should be fortified or undergo managed retreat should account for the benefits of allowing mangroves to undergo natural landward migration under a rise in relative sea level. These benefits include the continued provision of valued services and products, including consumptive benefits, education and research, aesthetic and cultural benefits, and future values such as a mangroves future potential for tourism (Dixon and Sherman, 1990; Ramsar Bureau, 1998).

#### **7.2.2. 'No-Regrets' Reduction of Anthropogenic Stresses**

Promoting overall mangrove ecosystem health by reducing and eliminating non climate-related stresses, such as filling and pollution, will increase mangrove resistance and resilience to climate change. These “no regrets” mitigation actions are justified and beneficial even in the absence of adverse effects on mangroves from climate change, where mitigation actions are typically undertaken to address both climate and non-climate threats (Adger et al., 2007). The value of wetlands conservation is often underestimated, especially in less developed countries with high population growth and substantial development pressure, where short-term economic gains that result from activities that adversely affect wetlands are often preferred over the less-tangible long-term benefits that accrue from sustainably using wetlands. The status of mangrove wetlands as one of the most threatened natural communities worldwide, of which roughly 50% of the global area has been lost since 1900, with losses during the last quarter century ranging between 35 and 86% (Valiela et al., 2001; FAO, 2003; Duke et al., 2007), supports this observation. Stresses associated with relative sea level rise and other effects from climate change present one set of threats to mangroves and other coastal ecosystems. Mangroves experiencing stress from other anthropogenic activities, such as clearing trees and dumping of pollutants, will be less resilient to these additional climate-related stresses. Local communities and leaders must recognize the long-term benefits of

mangrove conservation if we are to reverse historical trends in loss of mangrove area. This will provide the needed support to maximize mangrove resistance and resilience to climate change, and enable unobstructed natural landward migration wherever possible.

To attempt to augment mangrove resistance to site specific relative sea-level rise, activities within the mangrove catchment can be managed to minimize long-term reductions in mangrove sediment elevation, or enhance the trend in sediment elevation. For instance, limiting development of impervious surfaces within the mangrove catchment, and managing rates and locations of groundwater extraction, can reduce alteration to natural groundwater recharge to the mangrove systems, which might be an important control on mangrove elevation (e.g., Whelan et al., 2005). Also, avoiding and limiting human activities that reduce mangrove soil organic matter accumulation, such as the diversion of sediment inputs to mangrove systems, nutrient and pollutant inputs into mangroves, and mangrove timber harvesting, can contribute to maintaining relatively natural controls on trends in sediment elevation. Continual enhancement of mangrove sediment accretion rates could theoretically be conducted to augment the mangrove sediment elevation rate (Lewis, 1990), while avoiding excessive or sudden sediment deposition (Ellison, 1998).

### **7.2.3. Protected Areas**

Protected areas are one coastal resource management tool that can contribute to mitigating anticipated mangrove losses in response to climate change effects. Managers selecting sites and boundaries for individual protected areas, reviewing the effectiveness of existing protected areas, and designing protected area systems need to explicitly incorporate anticipated coastal ecosystem responses to climate change effects (Barber et al., 2004). For instance, planners need to account for the likely movements of habitat boundaries and species ranges over time under different sea level and climate change scenarios, as well as consider an areas' resistance and resilience to projected sea level and climate changes and contributions to adaptation strategies. Site-specific analysis of resistance and resilience to climate change when selecting areas to include in new protected areas should include, for example, how discrete coastal habitats might be blocked

from natural landward migration, and how severe are threats not related to climate change in affecting the site's health.

Protected area selection can include mangrove areas that act as climate change refugia, communities that are likely to be more resistant to climate change stresses. For instance, mature mangrove communities will be more resistant and resilient to stresses, including those from climate change, than recently-established forests. Protecting refugia areas that resist and/or recover quickly from disturbance in general, or that are predicted to be able to keep pace with projected relative sea-level rise can serve as a source of recruits to re-colonize areas that are lost or damaged (Palumbi et al., 1997; Bellwood and Hughes, 2001; Salm et al., 2006).

Ensuring representation of all mangrove community types when establishing a network of protected areas and replication of identical communities to spread risk can increase chances for mangrove ecosystems surviving climate change and other stresses. Ensuring that a portfolio of each different community type is represented is a strategy for optimizing climate change resilience as this representation increases the chance that at least one of these communities with disparate physical and biological parameters will survive climate change stressors and provide a source for re-colonizing (Julius and West, 2008). Replication, through the protection of multiple areas of each mangrove community type, by protecting multiple examples of each vegetation community (e.g., *R. mangle*-dominated communities, *B. gymnorhiza*-dominated communities, and stands of *X. granatum* in American Samoa) and geomorphic setting (e.g., back barrier, embayment, estuarine, deltaic, open coast, and drowned valley settings, Woodroffe et al., 2002), can help avoid the loss of a single community type (Roberts et al., 2003; Salm et al., 2006; Wells, 2006).

Furthermore, as a part of the recommended policy of adaptation to mangrove responses to relative sea level rise, the selection of sites for protected areas should account for connectivity or functional linkages between coastal ecosystems (Crowder et al., 2000; Stewart et al., 2003; Roberts et al., 2001, 2003). For instance, protecting a series of mature, healthy mangrove sites along a coastline could increase the likelihood of there being a source of waterborne seedlings to re-colonize sites that are degraded. Protected areas designed to preserve biodiversity and relatively

pristine habitats should incorporate adjacent coastal forests, mangroves, seagrass beds, and coral reefs to ensure all functional links are maintained in a least disturbed state. Protected areas designed in this manner will have optimal resistance and resilience to climate change and other stresses. The existence of functional links between coastal systems means that degradation of one habitat type will adversely affect the health of neighboring habitats. If a protected area encompassing a mangrove wetland does not include adjacent ecosystems, unsustainable activities occurring in the adjacent hinterland or offshore on adjacent coral reefs and seagrass beds could result in degradation of the mangrove (Barber et al., 2004; Ellison, 2004; Mumby et al., 2004).

#### **7.2.4. Mangrove Rehabilitation**

Restoring areas where mangrove habitat previously existed and creating new mangrove habitat where it did not previously exist (habitat conversion) will help offset anticipated reductions in mangrove area from relative sea-level rise. Enhancing degraded mangroves by removing stresses that caused their decline will increase their resistance and resilience to climate change (Hansen and Biringer, 2003; Ellison, 2004). Rehabilitation may be more successful and ecologically appropriate if mangrove wetlands are restored at sites where mangrove wetlands historically existed (Gilman, 1998; Kusler and Kentula, 1990; U.S. Environmental Protection Agency, 1993; U.S. Department of Defense et al., 1995; Erftemeijer and Lewis, 2000; Lewis et al., 2006). Appendix 1 describes principles and practices for mangrove rehabilitation and presents preliminary results from an American Samoa mangrove restoration initiative in order to demonstrate how the general principles and practices for mangrove restoration can be implemented.

#### **7.2.5. Monitoring**

The regional capacity assessment identified the need to establish a regional mangrove monitoring network, in part, to provide fundamental information to understand effects on mangroves from climate change (Section 6.3.6). Despite international recognition of this priority both at the United Nations level (UNEP/IOC/WMO/IUCN, 1991; Ellison, 1991, 1992; UNESCO, 1993) and within the region (South Pacific Regional Environment Programme,

1999a,b), the requisite technical and institutional capacity-building needs have yet to be addressed. This section identifies the parameters that need to be monitored and techniques employed to conduct the monitoring for effective identification of global climate change effects on mangroves.

While these techniques alone would not make it possible to determine causes of observed changes in mangroves, as is possible through controlled experiments, regional standardized monitoring may provide the basis for strong inferences of causation by global factors versus local influences on mangroves (CARICOMP, 1998). For instance, coordinated observations of regional phenomena, such as a mass mortality event of mangrove trees, or trend in reduced recruitment levels of mangrove seedlings, might be linked to observations of changes in regional climate such as reduced precipitation.

Changes in relative sea level and other climate change effects are expected to alter mangrove position and area (distribution), structure, and performance of functions. Table 7.2 identifies parameters that, if measured, would enable the establishment of mangrove baselines and detection of gradual changes, and explains how results of analyses of these monitoring data could be used to distinguish global from local factors inferred to be causing the observed changes.

Prescribed monitoring methods are also based on the assessment of technical, institutional and financial resources of the Pacific Island Countries and territories with native mangroves (Chapter 6). Detailed descriptions of most of the monitoring methods are found in Chapter 1 (Sections 1.5.6 and 1.5.7) and Chapter 3. Additional detailed descriptions of standardized monitoring techniques can be found in English et al. (1997), CARICOMP (1998) and Ellison and Oxley (1998). Tables 7.3 and 7.4 are used to implement some of the methods identified in Table 7.2.

Table 7.2. Parameters and monitoring techniques to establish baselines and distinguish global from local factors inferred to be causing observed changes in mangrove ecosystems.

Parameter	Monitoring Method(s)	How Analysis of Regional Database May Distinguish Global from Local Influences
Position of mangrove boundary	GIS analysis of (i) co-registered aerial photographs and satellite imagery; (ii) GPS surveys; and (iii) traditional survey techniques.	With combined information from these parameter, only from sites found to have a significant positive correlation between relative sea level rise and change in position of the mangrove boundary, will provide an improved understanding of migration rates of mangrove margins and change in profile in response to changes in relative sea level.
Change in sea level relative to the mangrove surface	(i) Analysis of tide gauge data; and (ii) Measurement of trend in change in elevation of the mangrove surface.	
Position of margins of vegetation zones	Establish a series of permanent parallel transects, generally located 90° to the coastline, of sufficient number and locations to characterize each site. Periodically (e.g., every 2 years) record the position of margins of each vegetation zone along each transect.	Is there consistency in observations of changes in mangrove zone boundaries for mangrove sites in the network that are experiencing similar global climate change effects? For instance, observations from mangrove sites included in the network may demonstrate consistent changes with rise, lowering and no change in relative sea level, confirming or denying hypotheses about mangrove responses to change in relative sea level. For instance, are the mangroves in the monitoring network that are experiencing a rise in sea level relative to the mangrove surface experiencing a landward migration of mangrove zone boundaries, is there large inter-annual variability in this parameter, are trends apparent over longer time periods of decades?
Assessment of human impact.	At standardized locations (e.g., at the middle of each vegetation zone along each permanent transect) assess the degree of impact within a 15 m radius. Impact is assessed on a scale from 0 to 5 where 0 is no impact and 5 is severely impacted (Tables 7.3 and 7.4).	Is there consistency in observations of changes in these three parameters for mangrove sites in the network that are experiencing similar global climate change effects? For instance, do sites experiencing increased precipitation consistently demonstrate increased health (lower degree of impact, increase in canopy cover and increase in tree seedling growth and productivity)? Or, for example, do sites experiencing a rise in relative
Canopy percent cover	At standardized locations (e.g., at the middle of each vegetation zone along each permanent transect) measure tree canopy cover within a 15 m radius using a spherical crown densitometer.	

Mangrove community structure, tree and seedling growth, and biomass	Establish permanent plots in each vegetation zone, of sufficient number and locations to characterize each site. Avoid unique spots, such as next to a tidal creek or development. For each tree and sapling in the plot, hammer in a tag at roughly 1.3 m height using a stainless steel nail and numbered tag. For trees too small to install a nail, attach the tag with a loop of stainless steel wire and clasp this onto a suitable low branch. Measure the tree diameter 2 cm above the tag. Within each permanent plot, count the number of seedlings (< 1 m in height) for each species.	sea level demonstrate decreased health (higher impact, reduced canopy cover, low degree of tree and seedling growth and productivity) along their seaward margins? Are there consistent observations region-wide of phenomena such as mass mortality of mangrove trees or reduced recruitment, which can be correlated to a factor such as decreased precipitation?
Phenology and productivity	In each permanent plot hang replicated litter catchers each of 1 m <sup>2</sup> . Empty these each month, dry contents and sort into productive parts (leaves, wood, propagules etc.)	Annual productivity per m <sup>2</sup> of mangrove can be calculated from averaged results.

Table 7.3. Codes used to record the impact of pressure on mangrove ecosystems (adapted from Ellison and Oxley, 1998).

Code	Impact	% Cover Canopy	Example
0	No Impact	96-100	Even canopy of trees. No gaps. No evidence of human interference.
1	Slight Impact	76-95	Canopy of trees fairly continuous but some gaps. Some regrowth. Isolated cutting/ stripping of trees or some evidence of pigs digging up saplings.
2	Moderate Impact	51-75	Broken canopy of trees with lower regrowth and recruitment areas. Some trees cut and stripped.
3	Rather High Impact	31-50	Tree canopy is uneven, the majority of the area is not showing regrowth and there is bare mud.
4	High Impact	11-30	Only a few trees remain at canopy height. Extensive clearance and some recruitment, large areas of bare mud
5	Severe Impact	0-10	Extensive clearance to bare mud, little recruitment, few trees remain alive

Table 7.4. Codes used to describe the type of impact at a site (adapted from English et al., 1997, Table 3.5).

Code	Type of Impact
BU	Bunding or dyking
BS	<i>Bruguiera</i> stripping for tapa dyes
CO	Infrastructure including houses, piggeries, garbage dumps, jetties, fish landing sites, construction sites or other coastal developments
ER	Erosion- shown by uneven mud surfaces or little scarps/ cliffs
IC	Illegal cutting
MI	Mining activities such as sand collection
MU	Multiple impact. Note codes of multiple impacts in Remarks.
OT	Others eg. pig foraging. Note this in remarks.
PP	Prawn and fish ponds
SC	Shell collecting
SS	Severe storm

Development of a guidebook to describe the methods of the standardized mangrove monitoring and database management would facilitate regional consistency.

Linking local mangrove monitoring programs that employ these standardized techniques may enable the separation of site-based influences from global changes to provide a better understanding of mangrove responses to effects of global climate change (CARICOMP, 1998; Ellison, 2000; Nurse et al., 2001). Employment of standardized techniques is needed to enable a meaningful comparison of results from the different programs. Pacific Island Countries and territories could establish a centralized data management center to serve as a repository for monitoring data from sites in the network and provide permanent resources to conduct requisite analyses of the regional dataset to identify effects from global change as well as distribute results of analyses to participating sites on a regular basis (CARICOMP, 1998).

The centralized data management center could make raw data accessible from the internet. This regional program could also provide periodic training opportunities to establish mangrove monitoring sites, employ the standardized monitoring methods, manage the local monitoring database, analyze monitoring data, interpret results, and identify management implications.

This regional program might be established at the Secretariat of the Pacific Regional Environment Programme, University of the South Pacific, University of Guam, University of Hawaii or other regional organization.



#### **7.2.6. Outreach and Education: Examples from American Samoa**

Education and outreach programs can increase the likelihood of successful implementation of mangrove conservation initiatives, including implementing actions to adapt to mangrove responses to climate change effects. Education and outreach activities are an investment to bring about changes in behavior and attitudes by having a better-informed community of the value of mangroves and other ecosystems. This increase in public knowledge of the importance of mangroves provides the local community with information to make informed decisions about the use of their mangrove resources, and results in grassroots support and increased political will for measures to conserve and sustainably manage mangroves. Education programs are developed for specific target groups as well as the general public. Examples include developing education kits for tour operators; training school teachers; developing school curriculums or activity modules for students; constructing boardwalks and interpretive signs; disseminating management information via pamphlets, radio, newspaper, and television; developing educational videos; and directly involving the local community in monitoring (Gilman, 2002). Interpretive structures such as boardwalks and signs may be counterproductive if sustainable financing is lacking to enable ongoing maintenance.

To complement mangrove research and conservation activities, the American Samoa Coastal Management Program funded the design and installation of a mangrove viewing platform and signs, located in Lions Park overlooking Nu'uuli mangrove (Fig. 7.1). The Coastal Management Program also funded the production and local distribution of two posters on the mangroves of American Samoa (Figs. 7.2 and 7.3). There is optimism that making these materials available to the local community in American Samoa will augment support for mangrove conservation, including the implementation of land use planning rules that allow for mangrove landward migration in response to projected relative sea-level rise.

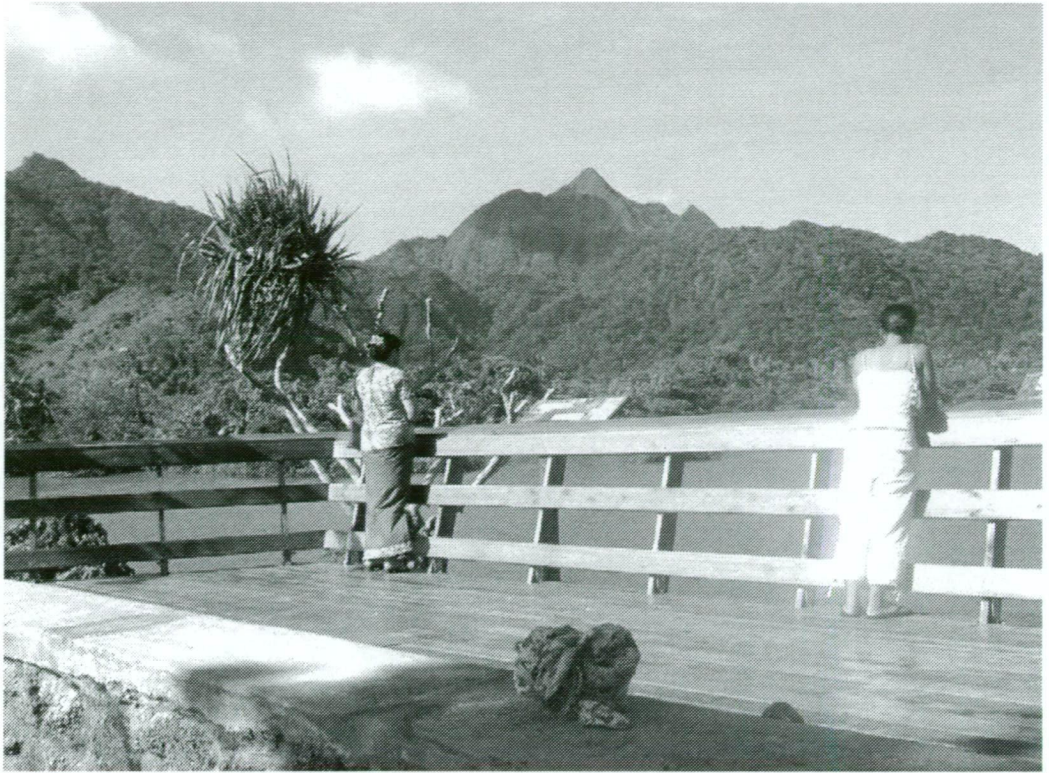


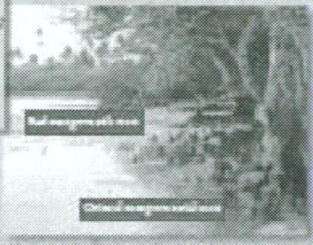
Fig. 7.1. Mangrove viewing platform and educational signs, Nu'uuli mangrove, American Samoa.



# Mangroves and the Samoan Way of Life



LIVING IN HARMONY WITH OUR MANGROVE WETLANDS



## MANGROVES IN AMERICAN SAMOA

The mighty Red Mangrove stands where dry land meets the sea, where the ocean and streams meet. The mangroves of American Samoa mangroves have been in existence for nearly 1000 years. Most mangroves were first planted in 1900. They are located in Tutuila and Apia's Marsh.

Two types of mangroves were introduced to American Samoa. The Coastal mangrove (Rhizophora mangle) is a tree that grows in salt water. The Red mangrove (Avicennia marina) is a tree that grows in brackish water. Both mangroves are important for the local economy and the environment.

Mangroves are a natural barrier against the sea. They are also a source of food and medicine.

## MANGROVES ARE WORTH PROTECTING

- **Coastal protection:** Mangroves protect our coastline from erosion and damage by waves. In fact, mangroves in American Samoa have saved us over \$7 million in damage from waves by the year 2000. Mangroves are also a source of food and medicine.
- **Wildlife habitat:** Mangroves are a natural habitat for many birds and fish. They are also a source of food and medicine.
- **Coral reef health:** Mangroves are a natural barrier against the sea. They are also a source of food and medicine.
- **Integral part of life:** Mangroves are a natural part of life in American Samoa. They are also a source of food and medicine.

Mangroves are a natural barrier against the sea. They are also a source of food and medicine.



Mangroves are a natural barrier against the sea. They are also a source of food and medicine.

## MANGROVE MONITORING AND REHABILITATION

The American Samoa Coastal Management Program, Land Use, and Wetland and Marine Living Resources Management Program, are working to monitor mangroves and protect them from damage. The program includes a survey of mangroves and a plan for their rehabilitation.

## THREATS

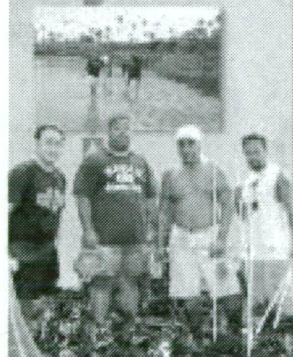
American Samoa mangroves face many threats, including logging, development, and climate change. Mangroves are a natural barrier against the sea. They are also a source of food and medicine.



Mangroves are a natural barrier against the sea. They are also a source of food and medicine.

## WHAT CAN YOU DO?

- Plant mangroves in your yard.
- Use mangroves for your garden.
- Use mangroves for your garden.
- Use mangroves for your garden.
- Use mangroves for your garden.
- Use mangroves for your garden.
- Use mangroves for your garden.
- Use mangroves for your garden.

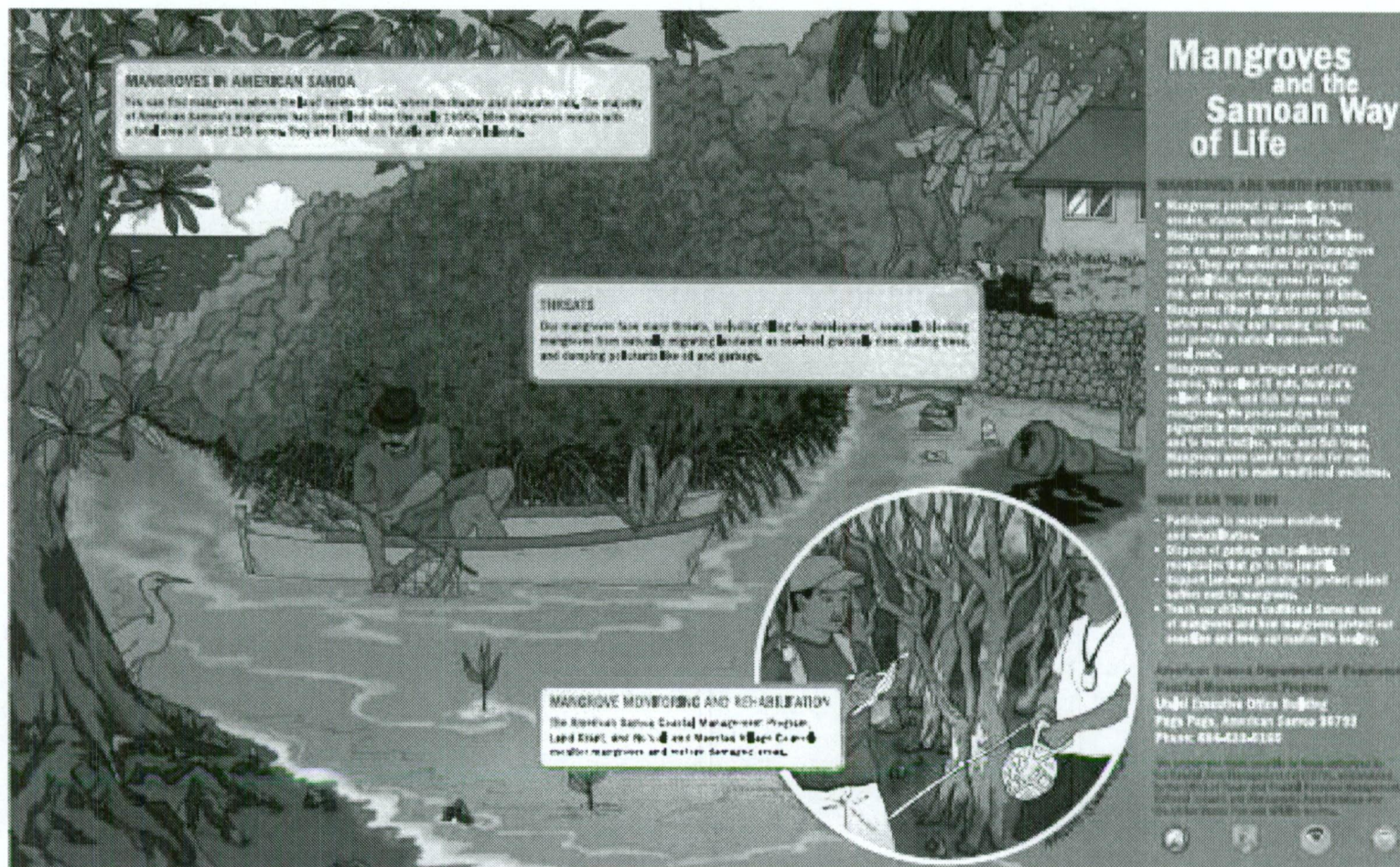


American Samoa Department of Government, Coastal Management Program, 1100 E. Highway 1, Apia, Samoa. Phone: (685) 497-1531. Fax: (685) 497-1531.













## Chapter 8

### Conclusions

#### 8.1. ASSESSING VULNERABILITY AND PREDICT RESPONSES TO RELATIVE SEA-LEVEL CHANGES

##### 8.1.1. Implications of American Samoa Findings and Research Needs

Results indicate that American Samoa mangroves are not likely keeping pace with rising sea-level, both surface and subsurface process controls on sediment elevation are important contributions to the vulnerability, and a large proportion of landward margins (16, 23 and 68 percent for the three sites) are obstructed from natural landward migration with sea-level rise. Based on observed trends in sediment surface elevations and movement of two mangroves' seaward margins, these sites have likely not been keeping pace with relative sea-level rise, with an elevation deficit of about  $2 \text{ mm a}^{-1}$  at both sites: Nu'uuli, an embayment mangrove, experienced sea-level rise relative to the mangrove sediment surface of  $2.22 (\pm 2.22 \text{ 95\% CI}) \text{ mm a}^{-1}$  and Masefau, a basin mangrove, experienced  $1.97 (\pm 0.32 \text{ 95\% CI}) \text{ mm a}^{-1}$ . At these sites, a highly significant positive correlation between the change in position of the seaward margins and change in relative sea-level suggests that rising sea-level relative to the mangrove surface caused the observed landward migration. Shoreline movement was not significantly correlated with changing sea-level at Leone, where development activities have likely been dominant factors determining changes in mangrove position. It is unclear if Leone has been keeping pace, as trends in sediment surface elevation were not monitored at this site.

The observed mean landward migration of three mangroves' seaward margins over four decades was 25, 64, and  $72 \text{ mm a}^{-1}$ , 12 to 37 times the observed regional relative sea-level rise rate. This is the first reported landward rate of migration of seaward mangrove margins, including at two sites where there is a significant positive correlation between the rate of margin migration and rate of relative sea level rise. Estimating the mangrove

seaward margin erosion rate using the Bruun rule, a predictive model of beach erosion (Bruun, 1962, 1988), resulted in inaccurate estimates. This is consistent with a large body of evidence that use of general predictive models such as the Bruun rule to estimate past erosion rates and future erosion estimates results in large error (SCOR Working Group, 1991; List et al., 1997; El-Raey et al., 1999; Pilkey and Cooper, 2004).

Where unobstructed by development, by the year 2100, the landward margins of the three mangroves are predicted to migrate landward 4, 12, and 14 m under an upper projection for change in sea level. The seaward margins are predicted to migrate landward 2, 27 and 30 m. The three mangroves could experience as high as a combined 47% reduction in area by the year 2100 as a result of the anticipated change in mangrove positions. There are no similar estimates in the literature for changes in individual mangrove site position and area resulting from relative sea level rise, based on assessments of physiographic settings, seaward margin erosion, change in mangrove sediment surface elevation, and change in relative sea level, as conducted here.

Reduced mangrove area and health will increase the threat to human safety and shoreline development from coastal hazards such as erosion, flooding, and storm waves and surges (e.g., Danielson et al., 2005). Mangrove losses will also reduce coastal water quality, reduce biodiversity, eliminate fish nursery habitat, adversely affect adjacent coastal habitats (Mumby et al., 2004), and eliminate a major resource for human communities that traditionally rely on mangroves for numerous ecosystem services (Section 1.2) (Satele, 2000).

Results from American Samoa are consistent with the literature, which documents that the majority of mangroves are not keeping pace with relative sea-level rise (Table 1.2) (Cahoon et al., 2006; Cahoon and Hensel, 2006; McKee et al., 2007). However, the mangrove sites where measured trends in sediment elevation have documented site specific relative sea-level rise should be considered preliminary. Longer study periods are needed to determine if these are long-term linear trends vs. cyclical, short-term patterns. Furthermore, these observations are primarily from mangroves of the



western Pacific and wider Caribbean regions. Assessments in other regions are needed to determine if this is a global vs. regional phenomenon.

Section 5.6 discussed the determination of a significant correlation between the historical observed rate of change in position of the mangrove margin and rate of change in relative sea level. This was conducted as part of the method to predict the future mangrove position, and employed methods used by Saintilan and Wilton (2001) and Wilton (2002). Due to a lack of information, neither trends in movement of mangrove landward margins nor net change in mangrove area were used to conduct this assessment. By default, trends in movement of the seaward margin were employed as an estimate of change in mangrove area. This may have produced unreliable results if landward mangrove margin positions had been controlled by factors in different magnitudes than the seaward margin.

Periodic mangrove boundary delineations would provide needed information on trends in mangrove area to enable a more reliable determination of the influence of sea-level trends on mangrove position, identified as a regional priority (Section 6.3.4). Alternatively, improvements in techniques for interpretation of historical remotely sensed imagery that enable reliable separation of mangrove from non-mangrove habitat would substantially improve methods for assessing mangrove vulnerability and predicting future margin positions. Improved technology in active remote sensing may eventually enable reliable separation of mangrove from non-mangrove signatures (Section 5.3). A time series of imagery using the new technology would be needed over a time period to provide a sufficiently long data series (years to decades) to determine a trend in mangrove margin position movement.

Predictions of changes in mangrove position over coming decades are also restricted, in part, due to the potential for numerous stressors that affect mangrove position to occur or change in magnitude during this time period (Sections 1.3, 1.5.3, 5.6, and 5.7). The occurrence and magnitude of several factors other than change in relative sea-level that will affect mangrove position (as well as structure and health) are difficult to predict, and potential effects on mangroves for many of these factors are not well understood. There is also limited understanding of the synergistic effects of multiple

climate change stressors and other anthropogenic and natural stressors on mangroves.

This was the first study of trends in extreme high waters for American Samoa. Changes in extreme high water levels and frequency were found to not pose an increasing threat to American Samoa mangroves beyond the effects from rising mean sea-level. Results are consistent with those of D'Onofrio et al. (1999), Woodworth and Blackman (2002, 2004) and Hunter (2002). This site-specific assessment supports the hypothesis that, in this region, which experiences large El Nino Southern Oscillation-related steric changes lasting several months to years, extreme high waters are likely to be related to mean sea-levels. Because there is spatial variability in trends in elevations and frequency of extreme high water events (Church et al., 2004b; Woodworth and Blackman, 2004), site-specific analysis is warranted, as conducted here for American Samoa. It is well understood how increases in extreme high water events threaten coastal development (Bijl et al., 1999; D'Onofrio et al., 1999; Hunter, 2002; Woodworth and Blackman, 2002, 2004; Church et al., 2001, 2004b). However, it is not well understood how changes in extreme high waters affect coastal ecosystems. Increased extreme high waters may have similar effects on mangroves as storms, including tree mortality and stress, sulphide soil toxicity, and altered sediment elevation through soil erosion, soil deposition, peat collapse, and soil compression (Stoddart, 1962; Craighead, 1964; Glynn et al., 1964; Lugo et al., 1976; Cintron et al., 1978; Smith et al., 1994; Mastaller, 1996; Woodroffe and Grime, 1999; McKee and Faulkner, 2000; Baldwin et al., 2001; Sherman et al., 2001; Woodroffe, 1995b, 2002; Cahoon et al., 2003, 2006; Cahoon, 2006; Piou et al., 2006).

### **8.1.2. Relative Threat from Sea-Level Rise**

To date, relative sea-level rise has likely been a smaller threat to mangroves than non climate-related anthropogenic stressors, such as filling and conversion for agriculture (IUCN, 1989; Primavera, 1997; Ramsar Secretariat, 1999; Smith et al., 2001; Valiela et al., 2001; Alongi, 2002), which have likely accounted for most of the global average annual rate of mangrove loss, estimated to be 1 to 2% (Valiela et al., 2001; FAO, 2003; Wells et al., 2006;

Duke et al., 2007). The estimate of the annual rate of mangrove losses in the Pacific Islands region in areas where mangroves are indigenous from 1980-2000 is 0.9%, slightly below the lower end of the global estimate (FAO, 2006). However, the validity of both estimates is questionable. Sea-level rise is predicted to be a substantial source of future mangrove losses locally and regionally. Losses in American Samoa could be as high as 47% by 2100. Regional extrapolation from the results in American Samoa predicted as high as 22.4% losses by 2100 (Section 3.10). Relative sea level rise will be a substantial cause of future reductions in regional mangrove area, causing about 25% of predicted annual regional losses and 11-22% of total estimated annual global losses (Section 5.8).

Direct climate change impacts (outcomes from changes in the atmosphere's composition and alterations to land surfaces) on mangrove ecosystems are likely to be less significant than the effects of associated sea-level rise. Rise in temperature and the direct effects of increased CO<sub>2</sub> levels are likely to increase mangrove productivity, change the timing of flowering and fruiting, and expand the ranges of mangrove species into higher latitudes. Changes in precipitation and subsequent changes in aridity may affect the distribution of mangroves. However, outcomes of global climate change besides sea-level rise are less certain (e.g., there is disagreement over whether the tropical Pacific will become more El Nino- or La Nina-like in response to increased greenhouse gas concentrations, Vecchi et al., 2008), and the responses of mangrove ecosystems to changes in these parameters are not well understood (Section 1.3). The understanding of the synergistic effects of multiple climate change and other anthropogenic and natural stressors on mangroves is also poor.

For example, a mangrove that is experiencing an elevation deficit to rising sea-level may be located in an area experiencing decreased precipitation, where groundwater extraction for drinking water is predicted to increase. The combined effect of just these three stresses on the mangrove could result in an accelerated rate of rise in sea-level relative to the mangrove sediment surface, and at the same time cause decreased productivity, resulting in highly compromised resistance and resilience to stresses from climate change and other sources. Models have not been

developed to predict the effects of multiple stresses such as described in this hypothetical example. There is an urgent need to test the hypotheses that have been advanced on the likely effects of global climate change on mangroves as there are many uncertainties and the effects are likely to be felt over a very long time scale.

### **8.1.3. Comprehensive Assessment over a Human Time Scale**

Combined, the research components conducted in American Samoa provided information required to comprehensively assess both mangrove vulnerability and predict change in position in response to projected relative sea-level rise, making this study the first of its kind. Furthermore, the study employed research methods suitable for employment in less developed countries, including being affordable and feasible for implementation by in-country staff (Sections 6.3 and 6.4), fulfilling Thesis Aim 1 (Section 1.6).

Comprehensive assessment of mangrove vulnerability and change in position in response to projected changes in relative sea-level was achieved through an assessment of:

- (i) Trends in mangrove sediment surface elevation, accounting for all controls on elevation through the entire sediment profile;
- (ii) Rate of change in regional relative sea-level;
- (iii) Projections for the rate of change in site specific relative sea-level, based on the previous two components combined with locally-adjusted projections for change in global sea-level;
- (iv) Trend in horizontal position of the mangrove seaward margin, and determination of whether or not the observed movement can be inferred to be caused primarily by changes in relative sea-level; and
- (v) The mangrove's physiographic setting (slope of land upslope and location of obstacles along the landward margin).

The information from these assessments enabled a determination of whether or not a mangrove site has been keeping pace with rising sea-level. The information also enabled a prediction of whether or not a mangrove site will keep pace over coming decades, and if it will not keep pace, what is the site's capacity to naturally migrate landward.

There is a growing body of literature documenting assessments of mangrove vulnerability to changes in regional relative sea-level (Table 1.2). These studies, however, have not provided information on site-specific resilience, including how a mangrove's physiographic setting will affect future position and area.

There is also an extensive body of literature conducting paleoenvironmental shoreline reconstructions to establish the long-term response of mangroves to past sea-level fluctuations (Table 1.2, e.g., Sugimura et al., 1988; Woodroffe et al., 1985; Ellison and Stoddart, 1991; Woodroffe, 1992, 1995; Shaw and Ceman, 1999; Berdin et al., 2003; Ellison 1993, 2000, 2006). However, these studies do not support predictions of future changes over relatively short, human time scales due to the effects of modern anthropogenic stressors, and because paleoshoreline reconstructions do not provide the resolution needed to understand mangrove responses to sea-level changes over a short time scale of decades.

There have also been numerous studies interpreting remotely sensed imagery and using a GIS to assess changes in mangrove and other habitat boundaries (e.g., Woodroffe, 1995; Solomon et al., 1997; El-Raey et al., 1999; Wilton and Saintilan, 2000; Saintilan and Wilton, 2001), as well as various vulnerability assessments based on simplistic comparisons of elevations of coastal ecosystems to predict future inundation based on projected changes in sea-level, and in some cases, the presence of obstacles to landward migration of coastal ecosystems (Gilman, 1990; Woodroffe, 1995; Solomon et al., 1997; El-Ray et al., 2003). Some studies have attempted to account for ecosystem responses to outcomes from climate change by employing the Bruun rule to estimate erosion. This likely has resulted in poor results relative to those that can be achieved through site-specific assessment, as conducted here (Section 5.4) (Bruun, 1988; List et al., 1997; Komar, 1998; Pilkey and Cooper, 2004).

## **8.2. GEOMORPHIC SETTING AND DEGREE OF CATCHMENT DISTURBANCE AS INDICATORS OF MANGROVE VULNERABILITY TO SEA-LEVEL RISE**

Given the current state of knowledge, understanding a mangrove's geomorphic setting does not negate the necessity to conduct site-specific monitoring to determine vulnerability to changing sea-level. The understanding of mangroves as opportunistic colonizers with distribution controlled through ecological responses to environmental factors (Tomlinson, 1986; Naidoo, 1985, 1990; Duke, 1992; Wakushima et al., 1994a, 1994b; Duke et al., 1998) highlights the importance of the geomorphic setting in determining where mangrove ecosystems establish, their structure and functional processes (Woodroffe, 2002). In theory, an understanding of a mangrove's geomorphic setting, including sedimentation processes (sediment supply and type), hydrology, and energy regime, should provide a basis for predicting vulnerability and responses to changes in sea-level, as these environmental parameters affect both surface and subsurface controls on elevation of the mangrove sediment surface. However, there have been no positive correlations observed between mangrove sediment elevation change and sedimentation rates (this study, and SET-MH studies, Cahoon and Hensel, 2006; Cahoon et al, 2006), regional relative sea-level rise, tidal range or soil bulk density (Cahoon and Hensel, 2006). Correlations have not been found between geomorphic classes and trends in mangrove sediment elevation in a global review conducted by Cahoon and Hensel (2006). This study has contributed information on the relative vulnerability of two mangrove geomorphic classes to relative sea-level rise, and the relationship between seaward mangrove margin erosion and relative sea-level rise, which can contribute to the eventual development of reliable predictive models for mangroves.

There was a highly significant difference in the mean sediment elevation change between the American Samoa embayment fringe site (Nu'uuli) and the estuarine basin site (Masefau), where the change in elevation was significantly lower at the embayment fringe mangrove, supporting the hypothesis that mangroves in an estuarine/drowned river valley composite geomorphic setting are more resistant to relative sea-level

rise than embayment mangroves (Thesis Aim 1, Hypothesis 1 supported). Mean landward migration of the mangroves' seaward margins was 12 to 37 times the relative sea-level rise rate. This is the first documentation of significantly different mean sediment surface elevation change for mangroves in different geomorphic settings, and the first documentation of the relationship between the rate of seaward mangrove margin erosion and relative sea-level rise rate, information needed to develop reliable predictive elevation models for mangrove ecosystems. Models used to predict salt marsh elevation responses to sea-level rise projections (Morris et al. 2002, Rybczyk and Cahoon 2002) have not been developed for mangrove systems, although Cahoon et al. (2003) has begun the process by developing a model for peat collapse after a hurricane. Until reliable predictive elevation models are developed for mangrove ecosystems, site-specific monitoring is necessary to assess mangrove vulnerability to sea-level rise.

Results, however, did not support the hypothesis that mangroves located in relatively undisturbed catchments would have a significant positive correlation between the rate of movement of mangrove margins and rate of change in regional relative sea-level, while mangroves in relatively disturbed settings would not demonstrate this correlation (Section 5.6) (Thesis Aim 1, Hypothesis 2 not supported). While a sample size of three mangrove sites is too small to base a definitive conclusion, results suggest that the proportion of development adjacent to the landward margin is not a good indicator of whether or not change in relative sea-level has been a primary control on mangrove position.

There is a need to develop models for reliable predictions of mangrove sediment elevation trends and elevation responses to sea-level rise projections, as has been done for salt marshes (Morris et al. 2002, Rybczyk and Cahoon 2002). Improved understanding of the linkages between different geomorphological classes and ecological and physical processes, necessary for the development of these mangrove predictive models, would facilitate improved predictions of the vulnerability to and responses to changes in sea level, and to identify adaptation methods to augment resistance and resilience. Improved understanding of how surface and subsurface processes control mangrove sediment surface elevation (Section

1.5.4), and feedback mechanisms resulting from changes in relative sea-level (Section 1.5.5) are necessary to develop reliable predictive models. For instance, it is necessary to understand how large an effect increasing sedimentation rate with rise in site-specific relative sea-level has on a mangrove site's ability to keep pace with relative sea-level rise (Cahoon and Hensel, 2006; Pethick, 1980, Saad et al., 1999; Allen, 1990a). There is also a need to improve the state of understanding of the effects of the range of land use practices on processes controlling mangrove sediment surface elevation (Cahoon and Hensel, 2006). For instance, the effects of chemicals in agricultural runoff on mangrove soil organic matter accumulation are poorly understood (Feller et al., 2003; McKee et al., 2002, 2007), as are the effects of increased impervious surfaces and concomitant reductions in groundwater recharge and increased surface water runoff on mangrove sedimentation and sediment elevation (e.g., Whelan et al., 2005). Observations over decades and longer and from numerous sites from a range of settings experiencing rise, lowering and stability in relative sea-level, may improve the understanding of short-term observations of positive correlations between relative sea-level rise and mangrove sediment accretion (Cahoon and Hensel, 2006) and other hypothesized feedback mechanisms (Section 1.5.5).

### **8.3. ASSESSMENT OF TRENDS IN MANGROVE SEDIMENT SURFACE ELEVATION**

#### **8.3.1. Full Sediment Profile**

Subsurface processes can be a substantial control on sediment elevation, as documented in this and previous studies (Krauss et al., 2003; Rogers et al., 2005a,b; Whelan et al., 2005; Cahoon et al., 2006; Cahoon and Hensel, 2006). As a result, it is necessary to observe trends in sediment elevation for full soil profiles (Fig. 1.8). Results from this study were consistent with the results of these previous studies, providing further support of the importance of monitoring the full soil profile to accurately measure trends in mangrove surface elevation: There was an estimated  $-4.05$  and  $-3.3 \text{ mm a}^{-1}$  of lowering in sediment elevation due to deeper subsurface processes in the two



American Samoa mangrove sites, offsetting shallow subsurface and surface processes causing a rise in sediment elevation of 3.8 and 3.3 mm a<sup>-1</sup>, resulting in net elevation lowering of -0.25 and 0 mm a<sup>-1</sup> in Nu'uuli and Masefau, respectively (Thesis Aim 1, Hypothesis 3 supported).

### **8.3.2. Pros and Cons of Stakes versus SET Methods**

In an attempt to identify an affordable and relatively simple yet scientifically rigorous method for the assessment of mangrove vulnerability to relative sea-level rise, this study was the first to employ stakes to observe trends in sediment elevation through sampling the full sediment profile (Fig. 1.8). Both the SET and stakes methods provide requisite information on trends in elevation to assess mangrove vulnerability to sea-level rise. Relative to the SET method, stakes are inexpensive: A SET costs USD \$1,000 and each benchmark USD \$500 (personal communication, Dr. Don Cahoon, U.S. Geological Survey, 1 October 2007). Depending on the depth of the mangrove sediment, in American Samoa, stakes cost about USD \$10 per 2 m length of pipe. To employ the sampling design in Nu'uuli and Masefau mangroves with combined 330 sampling sites, stakes cost about USD \$3,300. In comparison, the SET method would have cost USD \$166,000. The SET method has the added expense of requiring outside technical assistance for installation and initial training to conduct monitoring. The stakes method does not require outside technical expertise in most locations in the Pacific islands region to train local counterparts to conduct the installation and monitoring techniques (Table 6.1, 9 of 10 countries and territories have staff qualified to conduct mangrove surveys and inventories). Stakes are an affordable and less technically demanding alternative to the SET method, making it a more suitable method for broad application in less developed countries. In the Pacific Islands region, stakes are likely a better approach for use in a regional monitoring network, because local implementation of monitoring will be more likely to occur the less expensive the monitoring technique and the lower the demand for outside technical assistance.

SET-MH technology enables separation of various factors controlling sediment elevation (Section 1.5.6) (Boumans and Day, 1993; Cahoon et al., 2002a,b; Cahoon and Hensel, 2006). When used alone, stakes do not

capture information on the relative contributions of different portions of the sediment profile or individual factors when the stakes are all inserted through to the mangrove basal sediment horizon. Stakes could be used in combination with soil marker horizons, as typically conducted with studies employing the SET method. Multiple lengths of stakes also be used at individual sampling sites to calculate shallow subsidence or expansion over different depths of the soil profile, as is feasible with the SET method (Cahoon and Hensel, 2006).

Due to the lower cost per stake relative to a SET station, when funding is a limiting factor, the stake method enables a substantially larger sample size to better characterize a mangrove site. Observations of disparate trends in sediment elevation within individual mangroves (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007) highlights the benefit of a sampling design that includes a large number of monitoring locations. Employment of a large sample size makes the monitoring method more resistant to disturbance such as from human vandalism, as loss of individual stakes would not diminish statistical vigor when tens to hundreds of stakes are included in a single site, and the cost and effort to replace stakes is nominal.

Despite the advantages of the use of stakes over SET especially in less developed countries, there is an existing, extensive and growing network of coastal wetland sites employing standardized SET-MH technology, primarily at sites in the western Pacific and Wider Caribbean regions (Cahoon et al., 2006; Cahoon and Hensel, 2006). It may prove to be a more realistic option to expand the ad hoc SET network and transform and expand it into a formal network employing standardized monitoring techniques instead of introducing a new method.

### **8.3.3. Spatial Sampling Design and Study Period**

This study was the first to employ broad spatial coverage within and across different vegetation communities and localized geomorphic settings and use a relatively large number of sampling locations (330 in two sites) to observe trends in the elevation of two mangroves' sediment surfaces. This is a necessary spatial sampling design to adequately characterize mangrove sites, based upon evidence of high spatial variability in trends in mangroves'

surface elevation (Krauss et al., 2003; Rogers et al., 2005b; McKee et al., 2007) (Section 1.5.2.6).

Based on the results from this study, when this adequate, broad spatial sampling design is employed, a study period of decades, or at least several years, is likely required to distinguish statistically significant linear trends in the sediment surface elevation of coastal wetlands. To conduct an accurate vulnerability assessment of coastal wetlands to changes in site specific relative sea-level, monitoring must be conducted over periods that span the range of variability in surface and subsurface processes that control the elevation of wetland sediment surfaces. For instance, precipitation, evapotranspiration, river discharge, groundwater inputs, storms, and other factors causing short term pulses in sediment accretion and erosion and groundwater, are examples of processes that affect coastal wetland sediment elevation, which exhibit short-term, serially-correlated, variability. This suggested study period is necessary to differentiate between long-term linear trends and cyclical (including seasonal) trends (French and Stoddart, 1992; Kirby et al., 1993; Semeniuk, 1994; List et al., 1997; Reed et al., 1999; Krauss et al., 2003; Cahoon et al., 2006).

Alternative currently available methods to assess trends in mangrove sediment surface elevation all have limitations (Sections 1.5.6; 5.5.1, 8.3.2). The SET method (Boumans and Day, 1993) would be cost prohibitive to include a large number of sampling locations within a site, which is necessary to adequately characterize a large mangrove site with multiple geomorphic settings. Improved differential GPS technology may eventually become available for vertical measurements, enabling much more efficient monitoring of trends in sediment elevation in mangroves with a large number of monitoring locations relative to the SET and stakes methods (Section 5.5.1).

#### **8.4. AMERICAN SAMOA ADAPTATION PRIORITIES**

Results from the vulnerability assessment of American Samoa mangroves document site specific relative sea-level rise is very likely to be occurring in Nu'uuli and Masefau mangroves. The mangroves are predicted to continue to migrate landward over coming decades, where unobstructed. A large

proportion of all three mangrove study sites' landward margins contain obstacles to landward migration. For some portions of the coastline adjacent to these mangrove sites, managed retreat may be feasible, an adaptation strategy that could reduce anticipated mangrove losses (Section 7.2.1). For development located adjacent to the mangroves, especially for Leone and Masefau mangroves which have relatively good availability of upland behind existing coastal development, implementation of managed retreat may be a realistic option. This would reduce the proportion of the landward mangrove margins that are currently obstructed from landward migration. The American Samoa government could consider increasing the current buffer requirement for new development from wetlands, where setback distances adjacent to coastal ecosystems could be designed based on predicted landward migration rates. Rules regarding the construction of new and maintenance of existing seawalls and other hard engineering erosion control structures could be designed to implement a managed retreat policy as coastal ecosystems gradually migrate landward. Legal tools, such as rolling easements described by Titus (1991), could make eventual retreat and abandonment more acceptable.

Furthermore, observations of site specific relative sea-level rise in Masefau and possibly in Nu'uuli support implementation of no-regrets adaptation strategies in American Samoa to avoid and minimize anthropogenic stressors that contribute to reducing the mangrove sediment surface elevation (Section 7.2.2). For example, limiting development of impervious surfaces within the mangrove catchment and managing rates and locations of groundwater extraction could help to avoid and minimize alteration of natural groundwater recharge to the mangrove, which might be an important control on mangrove elevation (Whelan et al., 2005). Avoiding and limiting human activities that reduce mangrove soil organic matter accumulation, such as the diversion of sediment inputs to mangrove systems, nutrient and pollutant inputs into mangroves, and mangrove timber harvesting, can contribute to maintaining relatively natural controls on trends in sediment elevation (Feller et al., 2003; Cahoon and Hensel, 2006; Cahoon et al., 2006; McKee et al., 2002, 2007).

Additional adaptation approaches, described in Chapter 7, will be beneficial in American Samoa. Mangrove restoration, conducted in a pilot study in Nu'uuli mangrove (Appendix 1), and standardized monitoring have begun in American Samoa, two adaptation methods that are also regional priorities.

## **8.5. REGIONAL CAPACITY ASSESSMENT IMPLICATIONS AND ADAPTATION PRIORITIES**

A regional assessment documented low capacity in the Pacific Islands to assess mangrove vulnerability and low preparedness to adapt to mangrove responses to climate change (Thesis Aim 2, Hypotheses 1 and 2 supported). Regionally, there is a lack of information from site vulnerability assessments, and lack of experience incorporating information from site-based vulnerability assessments into land use planning processes. Adaptation plans adopted to date are simplistic and overly general, and are not based on empirical evidence from site-based assessments, due to a lack of rigorous site-based vulnerability assessments. Examples of some of the general adaptation strategies to manage mangrove and other coastal ecosystem responses to climate change effects identified in several of the nations' National Communication reports to UNFCCC are summarized in Section 6.2. There have been some national and island-scale qualitative vulnerability assessments in Pacific Island Countries and territories, which provide generalized predicted responses of coastal systems to projected change in sea level and other outcomes of climate change (e.g., Solomon et al., 1997; Phillips, 2000).

Several regional technical and institutional capacity-building priorities were identified. These include:

- Determining trends in relative mean sea level and trends in the frequency and levels of extreme high water events;
- Measuring trends in mangrove sediment surface elevation to determine how sea level is changing relative to the mangrove surface;
- Acquiring and analyzing a time series of remotely sensed imagery to observe historical trends in the position of mangrove margins;

- Producing topographic maps and maps of locations of development and roads for land parcels adjacent to and containing mangroves, and establishing or augmenting GIS programs. The World Bank-funded Infrastructure Asset Management Project in progress in Samoa might serve as a suitable regional model;
- To distinguish local and climate change effects on mangroves, establishing national standardized mangrove monitoring programs to comprise a regional mangrove-monitoring network. Providing training opportunities for in-country personnel to manage the mangrove-monitoring program, coordinate with a regional hub, and conduct monitoring techniques is further required in some locations. Monitoring would include periodic delineation of mangrove margins;
- Reducing stresses on and rehabilitating mangroves, in part, to increase their resistance and resilience to climate change; and
- Strengthening management frameworks to assess mangrove vulnerability and incorporate results into land-use plans.

Establishing a regional mangrove monitoring network may enable many of the identified regional capacity-building priorities to be fulfilled. Participating countries and territories could share technical and financial resources to maximize monitoring and conservation benefits through economy of scale. Establishment of a regional mangrove monitoring network has been recognized as a regional priority for about a decade (South Pacific Regional Environment Programme, 1999a,b), but little progress has been made (Section 6.3.6). A possible explanation for this lack of progress is the relatively low fundability of mangrove conservation activities, in particular, for monitoring activities. Pursuing the establishment of a regional coastal ecosystem (coral reef, mangrove, sea grass, beach, mudflat) monitoring programme, expanding upon successful coral reef monitoring networks, might be a more viable strategy. This approach would also optimize benefits from limited human and financial resources in the Pacific Islands region.

The research methods and results for site-based mangrove responses to relative sea level rise in American Samoa contribute to the understanding of how to design monitoring programs to assess mangrove responses to

changes in relative sea-level. However, there is substantially less understanding of how to monitor for mangrove responses to other climate change measures resulting from changes in the atmosphere's composition and alterations to land surfaces. There is also a gap in knowledge and experience on how to design and use information resulting from a mangrove monitoring network to separate local from regional and global influences (Sections 6.3.7, 7.2.5) (UNEP/IOC/WMO/IUCN, 1991; Ellison, 1991, 1992; UNESCO, 1993; CARICOMP, 1998; Ellison, 2000; Nurse et al., 2001). More research is needed on assessment methods and standard indicators of change in response to effects from climate change, as well as decision planning tools to incorporate information resulting from monitoring of mangrove responses to climate change.

Assessing the efficacy of legal and management frameworks to avoid and minimize adverse affects on mangroves and other valuable coastal ecosystems and plan for any landward mangrove migration is critical. Ensuring that legal and management frameworks are capable of eliminating and minimizing stresses that degrade mangroves is necessary to provide for mangrove resistance and resilience to anticipated stresses from sea level and other climate change effects. Managers will also require the institutional capacity to plan for site-specific mangrove responses to climate change, such as instituting setbacks from mangroves for new development for appropriate sections of coastline. However, management frameworks will only be effective if local communities and management authorities recognize the value of mangrove conservation. It is therefore also a priority to continually develop, augment and maintain a mangrove conservation ethic.

While, in concept, instituting alternative adaptation strategies to address scientifically-based advice should be straightforward, there has been no experience in doing so in the Pacific Islands region. If provided with site-based information on mangrove vulnerability to climate change, it is unclear whether authorities possess the political will to modify rules governing land use to achieve the long-term benefits from sustainable development and conserving coastal ecosystems. This is especially the case in many Pacific Island countries and territories where population growth is high and land suitable for development is limited. In many parts of the Pacific Islands

region, it will be particularly challenging to retreat from a landward-migrating coastal ecosystem or to establish zoning setbacks for new development. Increased efforts to raise local community awareness of the values of coastal ecosystems, including mangroves, may help authorities receive public support to make difficult land use planning decisions to plan for climate change.

The value of wetlands conservation is often underestimated, especially in less developed countries with high population growth and substantial development pressure. Instead, short-term economic gains that result from activities that adversely affect wetlands are often preferred over the less-tangible long-term benefits that accrue from sustainably using wetlands. The status of mangrove wetlands as one of the most threatened natural communities worldwide supports this observation: Roughly 50% of global mangrove area has been lost since 1900 (Ramsar Secretariat, 1999). Globally, losses continue at about 1-2% annually, with losses during the last quarter century ranging between 35 and 86% (Valiela et al., 2001; FAO, 2003; Wells et al., 2006; Duke et al., 2007). Stresses associated with relative sea level rise and other effects from climate change present one set of threats to mangroves and other coastal ecosystems. Mangroves experiencing stress from other anthropogenic activities, such as clearing trees and dumping pollutants, will be less resilient to these additional climate-related stresses. Local communities and leaders must recognize the long-term benefits of mangrove conservation to reverse historical trends in loss of mangrove area, maximize mangrove resilience to climate change, and where sea level is projected to rise relative to mangrove surfaces, enable unobstructed natural landward migration wherever possible. Education and outreach programs are an investment to bring about changes in behavior and attitudes by having a better-informed community of the value of mangroves and other ecosystems. This increase in public knowledge of the importance of mangroves can provide the local community with information to make informed decisions about the use of their mangrove resources, and can result in grassroots support and increased political will for measures to conserve and sustainably manage mangroves.



Site-specific economic valuation of Pacific Island mangroves (Section 1.2) could be conducted. This would enable cost-benefit analyses to be conducted to determine how the economic value of mangrove ecosystem services and products compares to estimated costs of implementing vulnerability assessments and adaptation measures. Consideration should be given to the limitations of economic valuation methods, reviewed in Section 1.2. Site-based valuation is warranted, as the range of reported mangrove values is wide (Ramsar Secretariat, 2001; Wells et al., 2006). Section 6.4 documents the cost to implement the mangrove vulnerability assessment method conducted in American Samoa, and discusses financing options available to Pacific Island countries.

There is a need to better plan our responses to climate change impacts on mangroves, especially in its identification through regional monitoring networks, and coastal planning that facilitates mangrove migration with sea-level rise and incorporates understanding of the consequence of shoreline changes. Establishing regional monitoring networks, including augmenting the knowledge of how to design monitoring and assessment methods (Section 7.2.5), is a priority in order to improve the understanding of how mangroves respond to outcomes of climate change. There is a need for site-based assessment of mangrove responses to relative sea-level rise and other outcomes of climate change, which can be conducted by employing the methods used here in American Samoa. Coastal site planning is needed to facilitate mangrove migration with sea-level rise. Measures to mitigate anticipated mangrove losses from sea-level rise and other climate change impacts, and to increase mangrove resistance and resilience to climate change, need to be incorporated into land use planning programs. The resistance and resilience of mangroves to sea-level rise and other climate change impacts can be improved by better “no regrets” management of other stressors on mangrove area and health, strategic planning of protected areas including mangroves and functionally linked ecosystems, rehabilitation of degraded mangroves, and outreach and education directed at communities residing adjacent to mangroves (Chapter 7). The development of process and decision tools that incorporate biological, social and economic factors

related to mitigating anticipated mangrove losses from climate change would be of additional benefit to mangrove managers.

#### **8.6. RECAP OF CONTRIBUTIONS TO THE STATE OF KNOWLEDGE AND REGIONAL RESEARCH AND MANAGEMENT PRIORITIES**

Results from the studies conducted in American Samoa and regional capacity assessment improve the state of knowledge for assessing mangrove ecosystem vulnerability and predicting responses to relative sea-level rise by determining that:

- **Comprehensive assessment of vulnerability and responses:** The comprehensive assessment of site specific mangrove vulnerability and responses to relative sea-level rise, employing methods suited for employment in less developed countries, is feasible. This was the first comprehensive site specific assessment of both mangrove resistance (mangrove's ability to keep pace with rising sea-level without alteration to its functions, processes and structure) and resilience (mangrove's capacity to naturally migrate landward in response to rising sea-level, such that the mangrove ecosystem absorbs and reorganizes from the effects of the stress to maintain its functions, processes and structure) to sea-level rise;
- **Mangrove vulnerability and resilience to sea-level rise:** Two American Samoa mangrove study sites have likely not been keeping pace with relative sea-level rise, based on there being significant linear trends in landward migration of seaward margins and rise in sea-level relative to mangrove sediment surfaces. Due to their physiographic settings, including the presence of development obstructing a large proportion of the landward margin from naturally migrating landward, if the trends in sediment surface elevation adopted here are valid, local losses from relative sea-level rise will be substantial: With no acceleration in regional relative sea-level rise over the next century (2004-2100), there will be a net 11.7% reduction in area of the three American Samoa mangrove study sites. This increases to 47.3% when applying the IPCC upper projection;

- **Extreme high waters:** Changes in extreme high water event levels and frequency are not likely to pose an increasing threat to American Samoa mangroves beyond the effects from rising sea-level;
- **Relation between rates of landward migration and relative sea-level rise:** The observed mean landward migration of the American Samoa mangroves' seaward margins was 12 to 33 times the observed site-specific relative sea-level rise rate. This observation is the first reported landward rate of migration of seaward mangrove margins at sites where there was a documented significant positive correlation between the rate of margin migration and rate of relative sea level rise;
- **State of predictive models:** The Bruun Rule produced an inaccurate estimate of past erosion rates of the American Samoa mangrove seaward margins. Given this state of predictive models for mangrove ecosystems, site-specific monitoring is necessary to assess mangrove vulnerability and predict responses to change in sea-level;
- **Significant difference in change in sediment surface elevation by mangrove geomorphic class:** Sediment elevation change was significantly lower at an embayment fringe mangrove site compared to an estuarine basin site in American Samoa. This is the first reported documentation of significantly different mean sediment surface elevation change for mangroves in different geomorphic settings. This supports the hypothesis that mangroves in an estuarine geomorphic setting are more resistant to relative sea-level rise than embayment mangroves;
- **Predicted regional change in mangrove area resulting from rising seas:** Regional extrapolation from the American Samoa results suggests reductions in mangrove area of between 2.4% to 22.4% through the year 2100 will occur, based on relative sea-level trends continuing linearly, and employing the IPCC upper projection, respectively. Relative sea level rise may cause 25% of regional annual mangrove losses and 11-22% of global annual losses;
- **Method to monitor trends in mangrove sediment surface elevation:** Stakes were a cost-effective method to monitor trends in mangrove sediment surface elevation, recognizing that broad spatial sampling is needed for accurate characterizations of mangrove ecosystems. This is

the first study design to employ such broad areal sampling to monitor trends in mangrove surface elevation. High uncertainties in observed temporal trends in American Samoa mangrove sediment surface elevations suggest that observations over a period of decades, or at least several years, may be necessary to discern statistically significant linear trends in the sediment surface elevation of coastal wetlands. This long study period is likely necessary in order to span the range of variability in surface and subsurface processes that control the elevation of wetland sediment surfaces;

- **Surface vs. subsurface controls of mangrove sediment surface elevation:** Results suggest that subsurface processes contributed substantially to observed changes in sediment surface elevation of the two American Samoa mangrove study sites, consistent with observations from other regions. This documentation of large differences between trends in sediment accretion and trends in sediment elevation in mangroves demonstrates that it is necessary to monitor entire soil profiles to obtain accurate trends in elevation of mangrove sediment surfaces. Sedimentation accretion or erosion rates alone are not accurate indications of trends in wetland surface elevation;
- **Monitoring trends in surface process contributions to mangrove sediment surface elevation:** Due to the combined limitations of radionuclide analysis of shallow sediment cores from mangroves, including the potential of soil profile disturbance over past decades, low  $^{137}\text{Cs}$  concentrations in the band of 10-20° S. latitude, and high cost for analysis per single sampling point, alternative methods to measure trends in sedimentation rates in mangroves are recommended, such as using marker horizons. While information on trends in sediment surface elevation determined from observations of surface processes may help identify effective adaptation strategies, this information per se is not used to assess mangrove vulnerability or predict responses to relative sea-level rise;
- **Regional capacity and state of planning:** There is low capacity in the Pacific islands to assess mangrove vulnerability as well as low preparedness to adapt to mangrove responses to climate change.

Except for this study in American Samoa, there have been no assessments of site specific mangrove vulnerability. There has been no site specific planning for adaptation to mangrove responses to climate change in the Pacific Islands region.

The following is a list of priority research, monitoring and coastal planning activities to improve the understanding of mangrove vulnerability and responses to climate change, including sea-level rise. Priorities for identifying site-specific options for adaptation are also included:

- **Trends in mangrove margin horizontal position:** It is a priority to invest in improvements in remotely sensed imagery, and techniques for interpretation, to separate mangrove from non-mangrove forest cover at a precision and accuracy to detect changes in position of centimeters per year or better. This would increase the precision and accuracy of assessments of mangrove vulnerability to relative sea level rise. To this end, it is also a regional priority to periodically delineate mangrove boundaries in order to obtain information on trends in mangrove area and position;
- **Improved techniques to measure trends in coastal wetland sediment surface elevation:** Relative to currently available alternative methods to monitor trends in mangrove sediment surface elevation (stakes, SET-MH, surveying), improved differential GPS technology for vertical measurements would enable efficient monitoring and the inclusion of a larger number of monitoring locations;
- **Effects of land use practices on mangrove surface elevation:** There is a need to improve the state of understanding of the effects of the range of land use practices (e.g., effects of agriculture fertilizers in runoff, effects of impervious surfaces on groundwater recharge and surface water runoff) on processes controlling mangrove sediment surface elevation;
- **Regional mangrove monitoring network:** Establishing regional monitoring networks, and augmenting the knowledge of how to design monitoring and assessment methods, is a priority in order to improve the understanding of how mangroves respond to outcomes of climate change. Linking local mangrove monitoring programs that employ standardized

techniques may enable the separation of site-based influences from global changes to provide a better understanding of mangrove responses to effects of global climate change. More research is needed on standard indicators of change in response to effects from climate change, as well as decision planning tools to incorporate information resulting from monitoring of mangrove responses to climate change;

- **Regional capacity-building priorities:** Pacific Island Countries and territories need to strengthen their technical and institutional capacity to assess mangrove vulnerability, including by: Determining trends in relative mean sea level and frequency and elevations of extreme high water events; measuring trends in the change in mangrove surface elevation; determining trends in horizontal position of mangrove margins; augmenting GIS programs, including the production of layers needed for an assessment of a mangrove's physiographic position; locally implement regional, standardized mangrove monitoring; and strengthen management frameworks;
- **Plan ahead:** Locally in American Samoa, and regionally, there is a need for improved coastal planning for adaptation to mangrove responses to climate change. This includes instituting measures to facilitate mangrove migration with sea-level rise (e.g., managed retreat). No-regrets management of activities within mangrove catchments that affect long-term trends in the mangrove sediment elevation, better management of other stressors on mangroves, rehabilitation of degraded mangrove areas, and increases in systems of strategically designed protected area networks that include mangroves and functionally linked ecosystems through representation, replication and refugia, are additional adaption options;
- **Reliable predictive elevation models for mangroves:** There is a need to develop models for reliable predictions of mangrove sediment elevation trends and elevation responses to sea-level rise projections. Improved understanding of the linkages between different geomorphological classes and ecological and physical processes, necessary for the development of these mangrove predictive models, would facilitate improved predictions of the vulnerability to and responses to changes in

sea level, and to identify adaptation methods to augment resistance and resilience. Improved understanding of how surface and subsurface processes control mangrove sediment surface elevation, and feedback mechanisms resulting from changes in relative sea-level are necessary to develop reliable predictive models. Until reliable predictive elevation models are developed for mangrove ecosystems, site-specific monitoring is necessary to assess mangrove vulnerability to sea-level rise;

- **Global studies of trends in coastal wetland sediment surface elevation:** There is a global need for substantially longer time series in studies monitoring trends in mangrove and salt marsh surface elevations to demonstrate the existence of any long-term linear trends. Furthermore, large spatial sampling within individual sites is needed in order to provide results on trends in sediment surface elevation that accurately characterize entire coastal wetland sites. This requisite broad spatial sampling, in part, creates the need for long (years to decades) study periods. Furthermore, observations of trends in surface elevations are primarily from mangroves of the western Pacific and wider Caribbean regions; assessments in other regions are needed to determine if the preliminary determination that most mangrove sites are not keeping pace with relative sea-level rise is a global vs. regional phenomenon;
- **Effects of direct climate change outcomes:** More research is needed to better understand the responses of mangrove ecosystems to changes in direct outcomes of climate change, including rise in temperature, increased CO<sub>2</sub> levels and changes in precipitation;
- **Synergistic effects:** Research is needed to improve the understanding of the synergistic effects of multiple climate change and other anthropogenic and natural stressors on mangroves. Models could potentially be developed to predict the effects of multiple stresses on mangroves;
- **Extreme high waters trends:** American Samoa is in a region that experiences large El Nino Southern Oscillation-related steric changes lasting several months to years, and as a result, extreme high waters are likely to be related to mean sea-levels. However, because there is spatial variability in trends in elevations and frequency of extreme high water

events, site-specific analysis as conducted here for American Samoa is warranted;

- **Extreme high waters effects on mangroves:** Research is needed to improve the understanding of how changes in extreme high water levels and frequency affect coastal ecosystems, including mangroves;
- **Improved interdisciplinary collaboration.** The various research components required to comprehensively assess site specific mangrove resistance and resilience to climate change requires interdisciplinary expertise. The needed areas of expertise include: wetlands science and processes (in particular, boundary delineation, interpretation of remotely sensed imagery of wetlands, and monitoring trends in elevation of sediment surface); time series data analysis (tide gauge data series, mangrove sediment surface elevation data series); and GIS analytical techniques. Creating opportunities for collaboration between professionals in these fields, and developing graduate-level programs with interdisciplinary curricula and research, will improve future capabilities to conduct studies to assess ecosystem vulnerability, predict responses, and recommend adaptation measures for optimal preparedness for climate change.



## Chapter 9

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## **Appendix 1**

### **Offsetting Anticipated Mangrove Losses: Case Study of American Samoa Mangrove Restoration**

Management authorities are encouraged to rehabilitate mangroves as a 'no regrets' strategy to mitigate predicted mangrove losses resulting from relative sea-level rise and other climate change effects (Section 7.2.2). Restoring areas where mangrove habitat previously existed will help offset anticipated reductions in mangrove area from relative sea-level rise. Enhancing degraded mangroves by removing stresses that caused their decline will increase their resistance and resilience to climate change (Hansen and Biringer, 2003; Ellison, 2004).

This Appendix describes principles and practices for mangrove rehabilitation. Methods and preliminary results from a mangrove restoration project conducted in American Samoa are also presented in order to demonstrate how the general principles and practices for mangrove restoration can be implemented.

#### **A1.1. INTRODUCTION**

##### **A1.1.1. Mangrove Rehabilitation Principles**

Determining the stress or stresses that caused a mangrove to decline helps to identify the restoration or enhancement method to remove these stresses (Lewis, 2005). Many attempts to rehabilitate mangroves have not adequately considered the major factors that control mangrove ecosystem survival and health (Lewis et al., 2006). Too often the approach taken to restore or enhance a mangrove site is to plant mangroves, without first identifying if some stress that is inhibiting natural regeneration is still present, which usually results in low or no survival of the planted mangroves (Lewis and Streever, 2000; Lewis, 2005). Only when the availability of waterborne seedlings of mangroves from adjacent stands is blocked is planting

mangroves necessary to restore a degraded mangrove site. Mangrove wetlands can self-repair over a period of 15-30 years if hydrologic functions are intact and natural recruitment of mangrove seedlings occurs (Lewis, 2005), although planting mangroves after removing causes of decline can help expedite this recovery.

Rehabilitation sites must meet the environmental conditions (duration, frequency and depth of inundation; wave energy; substrate conditions; salinity regime; soil and water pH; sediment composition and stability; nutrient concentrations; elevation; slope, etc.), wave energy, substrate conditions, salinity regime, soil and water pH, sediment composition and stability, nutrient concentrations, elevation, slope, etc.) required by the true mangrove species indigenous to the area. While it may be feasible to establish mangrove vegetation at new sites where they had not previously existed (habitat conversion) (e.g., Choudhuri 1994; Sato et al., 2005), rehabilitation may be more successful and ecologically appropriate if mangrove wetlands are restored at sites where mangrove wetlands historically existed (Gilman, 1998; Kusler and Kentula, 1990; U.S. Environmental Protection Agency, 1993; U.S. Department of Defense et al., 1995; Erftemeijer and Lewis, 2000; Lewis et al., 2006). Restoring the full suite of functions performed by a natural, healthy, relatively undisturbed mangrove community likely requires a long time period and might require active management, for instance, to prevent the establishment of alien invasive species.

Some site preparation requirements for mangrove rehabilitation include (Smith III, 1987; Kusler and Kentula, 1990; Naidoo, 1990; Lewis, 1994, 2005):

- **Conservation ethic:** If the mangrove site is located in an area inhabited by people, augmenting or developing a mangrove conservation ethic by the local community can be critical to the success of any rehabilitation initiative. In most cases, direct human disturbance is the cause of the original mangrove degradation or loss.
- **Elevation:** If necessary, grade the mangrove sediment surface to the elevation that provides optimal hydrologic regime (period, frequency, and depth of inundation) for the targeted mangrove species. Riley and Kent



(1999) planted mangrove seedlings within translucent, 3.8 cm-diameter poly vinyl chloride pipes with a partial vertical slit, in part, to establish plants at elevations lower than where natural recruitment was occurring. If fill will be added or removed to achieve the target elevation and slope, the design and careful monitoring of the final target grade is critical to the rehabilitation project's eventual success;

- **Slope:** Gradual slope helps reduce erosion, filters runoff entering the wetland, and allows for surface drainage at low tide;
- **Tidal exchange and wildlife access:** It may be necessary for large mangrove rehabilitation sites to include drainage channels to simulate natural tidal creeks, providing requisite tidal exchange, salinity regime, and wildlife access;
- **Wave energy:** If necessary, install an offshore structure (e.g., breakwater, rock berm, jetty, dike, or submerged sandbar) if the rehabilitation site is exposed to too high a degree of wave energy;
- **Fertilizer:** Consider if the time-release of fertilizer (N is a nutrient limiting growth of halophytes in intertidal areas) is warranted;
- **Fencing and removal of loose debris:** If researchers determine that the area is at risk of disturbance from humans, pigs, dogs, etc., installing fencing around the perimeter of the restoration site can help avoid damage to the rehabilitation area. Also, if there are dead trees or garbage on the site, then these should be removed. Dead trees can become loose and roll with tides and waves, as can garbage and other loose debris, which can damage the rehabilitation area.

Typically, the requisite approach to restore a degraded or lost mangrove site entails reestablishing the elevation and slope that is optimal for target true mangrove species, which determine the hydrologic regime (period, depth and frequency of inundation), and further augmenting a mangrove conservation ethic by the local community. The likelihood of successfully rehabilitating a mangrove is typically highest for sites where disturbance has been recent and can be stopped, because alteration to the physical structure of the mangrove is likely to be minimal.

### **A1.1.2. Rehabilitation Purpose**

It is important to define the purpose of mangrove rehabilitation, as this controls the methods and materials to be adopted and development of performance standards and monitoring techniques (Gilman, 1999). The objectives of mangrove rehabilitation projects have included timber production or silviculture, enhancement of coastal protection, improved water quality, but most commonly it is for the objective of restoration of degraded areas to attempt to restore structure and functional performance to a least disturbed state (Field, 1998; Lewis and Streever, 2000; Lewis, 2005).

The purpose of conducting the American Samoa mangrove restoration project was to attempt to restore the mangrove to perform functions at similar levels as the mangrove wetlands located adjacent to the restoration project site. This mangrove restoration project was conducted primarily to serve as a model for replication at the other 15 Pacific Island Countries and territories where mangroves are indigenous. There is a need to demonstrate effective and affordable mangrove restoration techniques in the Pacific Islands region as there is a dearth of mangrove rehabilitation expertise in the region (Section 6.3.8). There has been limited activity in the region in rehabilitation of mangroves, with small-scale successful projects only recorded from Kiribati, Northern Mariana Islands, Palau, and Tonga and failed mangrove rehabilitation efforts in American Samoa and Papua New Guinea. The results of two additional rehabilitation efforts in Palau and Fiji are not known. This highlights the need for improved staff training, capacity building and information sharing.

The project was also conducted to achieve local benefits. These include returning valued ecosystem services to the section of coastline where the project site is located, augmenting in-country capacity to monitor mangrove health and conduct mangrove restoration, augmenting a mangrove conservation ethic by the local community, and reversing trends in loss of mangrove area and health in American Samoa.

The project also seeks to develop local capacity to rehabilitate mangroves at other sites in American Samoa, a skill that is currently lacking. There is a need for training and experience conducting mangrove rehabilitation by American Samoa wetland scientists. The American Samoa

Community College Land Grant Program, with assistance from staff from the American Samoa Coastal Management Program, conducted an unsuccessful attempt to restore mangroves to the Nu'uuli project site through raising mangrove seedlings at a nursery and transplanting the seedlings at the restoration site. None of the seedlings survived, likely because the restoration method did not identify the cause of the mangrove loss and remove this stress.

#### **A1.2. STUDY AREA**

The Coconut Point area of Nu'uuli mangrove was selected for the location of a mangrove restoration project (Fig. A1.1a). The center of the mangrove restoration study site is located at 14° 19' 04.2" S latitude, 170° 42' 09.2" W longitude, located in the southeastern part of Nu'uuli mangrove (Fig. A1.1b). The restoration site is approximately 1650 m<sup>2</sup>, 55 m long (parallel to the shoreline) and 30 m wide (from landward to seaward margins).

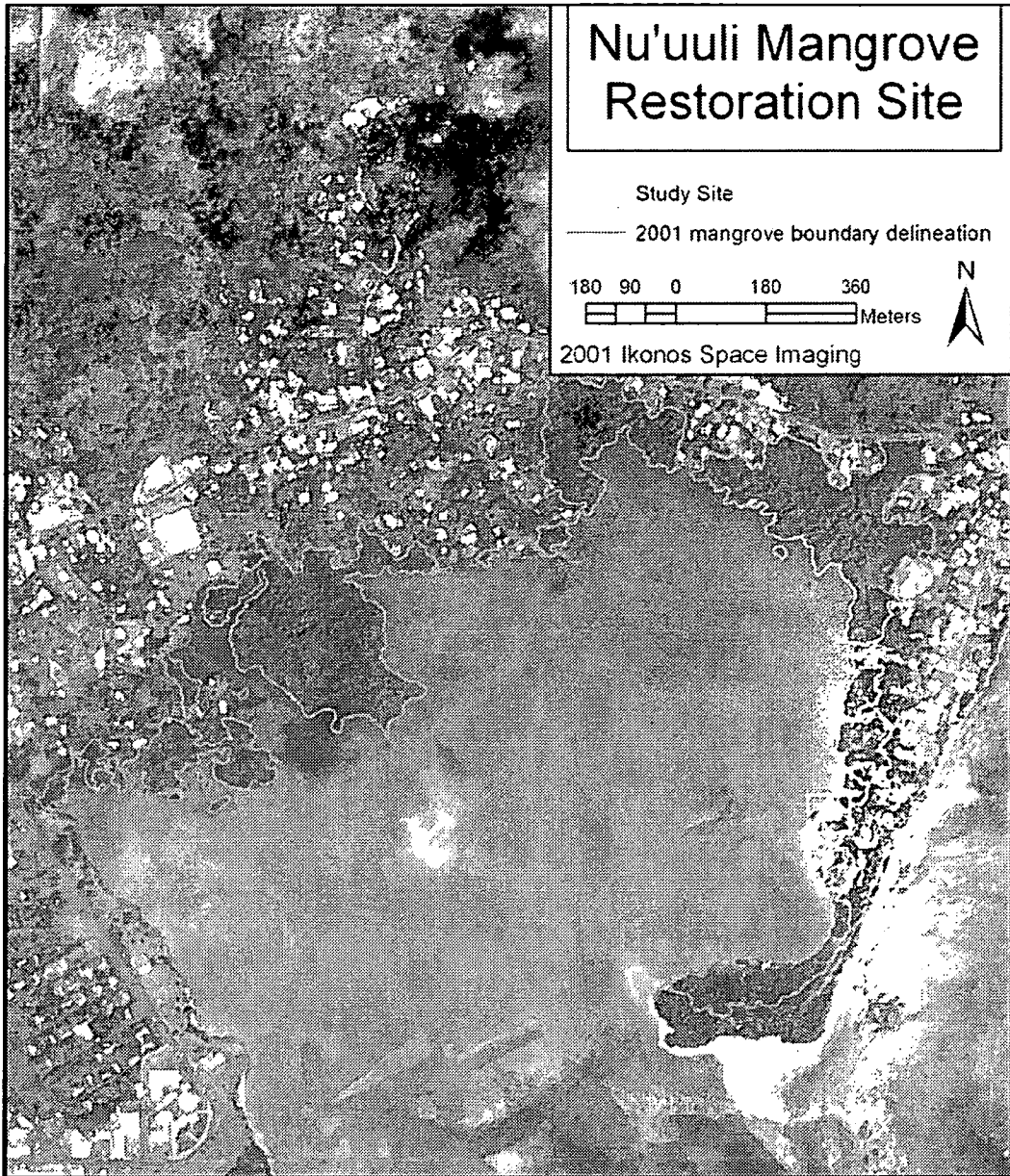


Fig. A1.1a. Location of Nu'uuli mangrove restoration site (blue square), showing the boundary of the entire mangrove site, Tutuila Island, American Samoa.

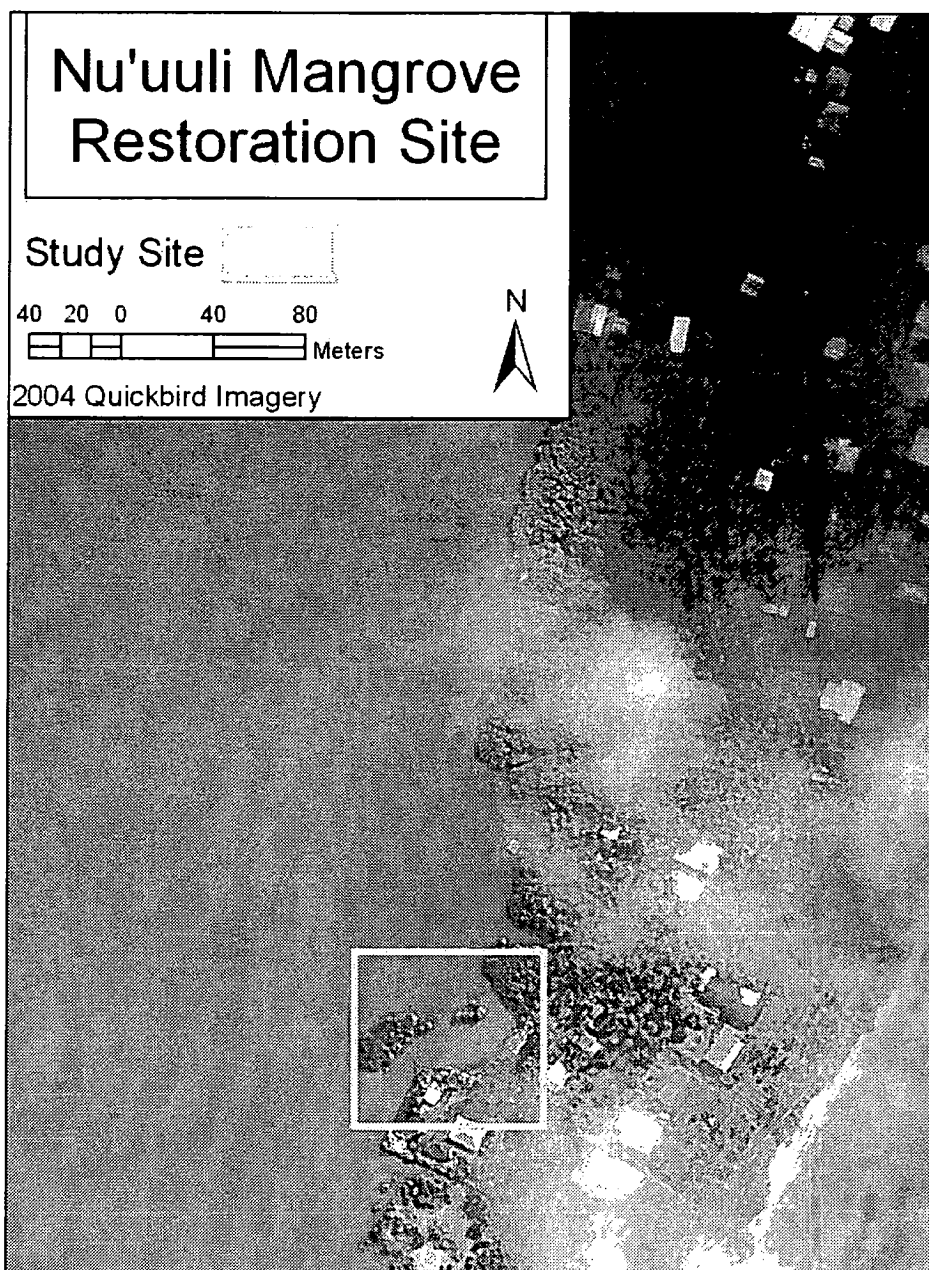


Fig. A1.1b. Nu'uuli mangrove restoration site, Tutuila Island, American Samoa.

### **A1.3. METHODS**

#### **A1.3.1. Period**

Initial restoration activities took place from 13-15 June 2006. Monitoring was conducted five times (13 July, 20 August, 5 October, 24 November and 8 December) over six months from initial restoration activities.

#### **A1.3.2. Site Selection and Community Participation**

The mangrove restoration site is located at Coconut Point, on the Southeastern edge of the largest mangrove area in the Territory of American Samoa in Nu'uuli mangrove of Pala Lagoon (Fig. A1.1a). This site was clear of mangroves apart from six mature *B. gymnorrhiza* trees. Very few seedlings and saplings are establishing despite an ample supply of propagules from adjacent relatively healthy mangroves. This site was selected for the project because it was determined that the system has been altered to such an extent that it could not self-correct (Fig. A1.2), the site has easy access making it convenient for training local agency staff to conduct mangrove rehabilitation and monitoring, and the site location makes it suitable for community participation in the restoration and monitoring activities. Furthermore, through pre-project community consultation it was determined that the adjacent landowners, village Mayor, village council and local community supported the project and members of the local community were available to participate in restoration and monitoring activities, and minimize the risk of human disturbance of the restoration site.



Fig. A1.2. Northern portion of the mangrove restoration site, Coconut Point, Nu'uuli Mangrove, Tutuila Island, American Samoa, before initiating restoration activities.

#### **A1.3.3. Determine Stresses that Caused Mangrove Decline**

A time series of aerial photographs showing Nu'uuli mangrove from 1961, 1971, 1984, and 1990 and 2001 Ikonos and 2004 QuickBird space imaging was analyzed to determine (i) when the mangrove vegetation disappeared from the site; (ii) if the mangrove vegetation at the restoration site has demonstrated any trend, such as a continual reduction in mangrove tree cover, which might indicate that erosion is occurring; and (iii) if the historical imagery provides information to support an inference of the cause of the loss of mangrove habitat from the restoration area. Section 3.3 provides information on the satellite imagery geo-referencing and co-registration of aerial photos used in this study.

The owners of the land parcels located immediately landward of the project site and the adjacent landowners were also interviewed to attempt to determine the cause of the mangrove loss.

#### **A1.3.4. Determine Target Substrate Elevation and Vegetation Zone Widths**

The historical aerial photos and satellite imagery were interpreted to determine the locations and widths of historical mangrove vegetation zones at the study site before the mangrove vegetation cover was lost. The width of the mangrove vegetation zones of an adjacent relatively healthy reference

mangrove site were also measured in order to help design the location of the restoration site mangrove vegetation zones.

The elevation of the mangrove of an adjacent reference mangrove site was compared to the elevations of the corresponding sediment surface of the restoration site to determine if disparities in elevation might be preventing natural regeneration, and to determine what elevation to target for the restoration site (Fig. A1.3).

This approach assumes that the restoration area is currently at a lower elevation than adjacent mangrove surfaces, and that by establishing mangrove trees the restoration site will gradually build up sediment to reach a surface elevation equivalent to that of the adjacent mangrove areas. Vegetational friction on water movement combined with flocculation of clays contributes to substrate accretion (Furukawa and Wolanski, 1996; Furukawa et al., 1997). Because propagules are present but not establishing at the study site, it is assumed that the disturbance stress that caused the mangrove loss or that is preventing natural regeneration is still present. Excavation of fill or backfilling of an excavated or eroded area to achieve the same slope and elevations relative to a reference site would be an optimal approach to achieve the correct hydrology (Lewis and Streever, 2000). However, this is expensive, and raises many additional complexities. For instance, fill material must be of suitable grain size and free of contaminants. The methods employed of using infilled tires to achieve target elevations and pipes to provide protection from debris and human disturbance, is a more cost-effective approach, the efficacy of which is being assessed in this study.





Fig. A1.3. Surveying elevations of the study site and adjacent *B. gymnorrhiza* and *R. mangle* zones of a relatively undisturbed reference mangrove site to determine the elevation deficit in the restoration area.

#### **A1.3.5. Restoring Mangrove Vegetation Zones**

In general, suitable species to be replanted are those that naturally occurred at the site before disturbance. Mangrove species tend to occur in zones according to micro-elevation and frequency of inundation. Therefore, it is best to replant with the species that used to grow in the zone, i.e. *R. mangle* on the seaward margin, and *B. gymnorrhiza* on the landward margin. Historical aerial photographs were analyzed to identify the former extent of mangroves and the constituent vegetation zones. The margins of a *B. gymnorrhiza* zone to be restored to the landward mangrove zone and width of the *R. mangle* zone to be restored along the narrow seaward fringe, and target density of trees, was determined based on the review of the available historical remotely sensed imagery and assessment of the widths of these mangrove zones in the mangrove wetlands adjacent to the mudflat area being restored.

Saplings were transplanted from wild sources. *B. gymnorrhiza* saplings were transplanted from a large supply from mangrove sites proximate to the restoration site within the Coconut Point area. Due to a difficulty in locating suitable *R. mangle* saplings for transplanting from

adjacent mangrove areas, about half of the restoration site *R. mangle* saplings were taken from an area in Nu'uuli mangrove outside of the Coconut Point area where there was a substantially higher soil organic content than at the restoration site. Mangrove saplings for replanting were collected from large, mature mangrove ecosystems where natural regeneration is occurring. An attempt was made to collect saplings only from areas within the forest with large populations of shaded saplings, and from areas where the mangrove mud was firm. This is because sediment is removed with the sapling, so in a narrow, degraded or sea margin areas, erosion and degradation of the source area may occur. Saplings were also not collected from light gaps as these saplings have a relatively high likelihood of surviving.

An attempt was made to choose saplings for transplanting that were 0.5 - 0.8 m tall, with a straight trunk, an intact growing tip, and several leaf pairs. An attempt was made to avoid old saplings, with over 15 leaf scars on the trunk (Duke and Pinzón, 1992), and those that already have developed prop roots or side branches. Older saplings are less likely to survive transplanting, probably due to root disturbance (Hamilton and Snedaker, 1984). Saplings were transported to the study site and transplanted within 30 minutes of being extracted from their original site.

Saplings were planted by digging and placing the sapling into a hole. An attempt was made to ensure that the mud level after planting was the same as at the original location. If the sapling were buried deeper it will likely not survive. Saplings were removed and holes were dug by hand, digging tools were rarely necessary.

Instead of relying on natural regeneration or planting propagules (the fruit after falling from the parent tree but not yet rooted in the substrate), saplings were transplanted in order to reduce the amount of time to restore the mangrove habitat to reference conditions and because use of tires with elevated sediment surface inside, in which planting occurred, prevents natural recruitment mechanisms. Saplings were not raised from seedlings in a nursery. Raising the seedlings in a nursery risks the occurrence of stress and low survivorship when the nursery-raised seedlings are transplanted to the project site: conditions (hydrologic regime, wave energy, salinity, nutrient

levels, sun exposure, etc.) in the nursery are likely to be very different from the project site. Advantages of wilding collection and transplanting are: saplings can be collected at any time through the year; they are suitable for higher energy sites; and success rates are usually higher than planting seeds.

Three treatments were employed to restore mangrove vegetation to the restoration site:

- **Rebar or other support structure adjacent to sapling:** A single *R. mangle* sapling, approximately three years old, was observed growing next to a 0.57 m tall pipe in the restoration study site (Fig. A1.4). This was the only *R. mangle* sapling present in the restoration site (there were also two existing *B. gymnorrhiza* saplings each about 1 year old, in the study site), supporting a hypothesis that sapling survival might be enhanced when located adjacent to similar support structures, perhaps by providing a degree of protection from human disturbance and debris. Based on this observation and hypothesis, one of the restoration method treatments was to place a 3 m length rebar pipe into the sediment adjacent to planted mangrove saplings. In some cases a pipe or wooden stick was used due to a shortage of locally available rebar;
- **Tires:** Used car and truck tires were laid flat on the sediment surface, filled with sandy sediment taken from close offshore from the restoration site, and one *R. mangle* or *B. gymnorrhiza* sapling was planted inside. The height of the sediment inside the tire was measured relative to the elevation of the sediment adjacent to the tire at project initiation and again at six months;
- **No physical support:** *R. mangle* and *B. gymnorrhiza* saplings were planted in the restoration site with no support structure placed within 1 m;



Fig. A1.4. A *R. mangle* sapling approximately three years old growing next to a pipe was the only sapling observed in the restoration study site.

Fig. A1.5 shows a plan view of the restoration site identifying the location and treatment for each transplanted sapling. The locations of the restored seaward *R. mangle* mangrove vegetation zone and landward *B. gymnorrhiza* zone are identified. Locations of existing and transplanted *B. gymnorrhiza* and *R. mangle* saplings, whether the sapling is situated inside a car tire, has a rebar pole or other structure adjacent to it, or has no support structure are also identified. The restoration project included the transplanting of 93 *B. gymnorrhiza* and 41 *R. mangle* saplings into the project site. Two pre-existing *B. gymnorrhiza* saplings were present in the restoration site (Z1 and L7, Fig. A1.5). One pre-existing *R. mangle* sapling was present (Z2, Fig. A1.5). Of these total 137 saplings, 52 (38.0%) were planted in tires, 68 (49.6%) had a rebar pipe or similar structure located adjacent to it, and 17 (12.4%) had no physical support structure. Of the total 95 *B. gymnorrhiza* saplings, 33 were inside a tire, 45 had a rebar pipe or similar structure located adjacent to it and 17 had no physical support. Of the total 42 *R. mangle* saplings, 19 were placed inside a tire and 23 had a rebar pile or similar structure located adjacent to it. Saplings were generally located at  $\geq 1$  m intervals, as this provides mutual protection. Also, planting seedlings in 1-m centers, or 10,000 per ha, will lead to the target tree density

of mature mangroves of about 1,000 trees per ha (1 tree per 10 m<sup>2</sup>) (Lewis and Streever, 2000).

Fig.A1.6 shows the restoration site at the time of completion of initial restoration activities. Codes for each sapling in the study site were initially painted on the outside of tires and written on flag tape, placed on pipes and saplings. Aluminum tree tags were later attached to each sampling with loosely fit wire.

The experimental design was purposely not randomized or balanced. A higher proportion of saplings with a physical support structure (adjacent to a pipe or inside a tire) versus saplings with no support was intentional, and the inclusion of a pipe or tire for all saplings in the seaward *R. mangle* zone where wave energy and exposure to impact from debris is highest was intentional, in an attempt to maximize the likelihood of sapling survival.

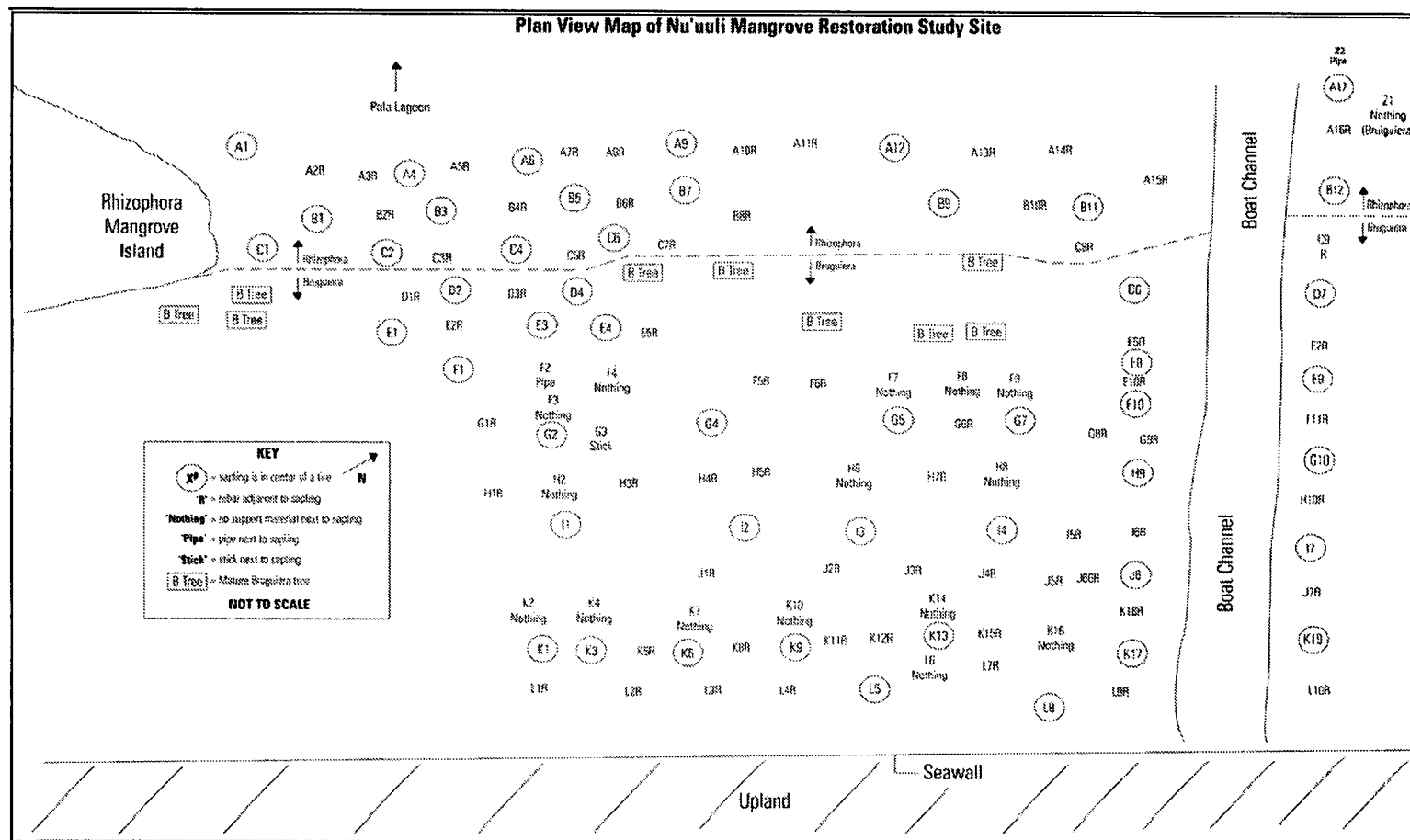


Fig. A1.5. Plan view of the Nu'uuli mangrove, American Samoa restoration study site.

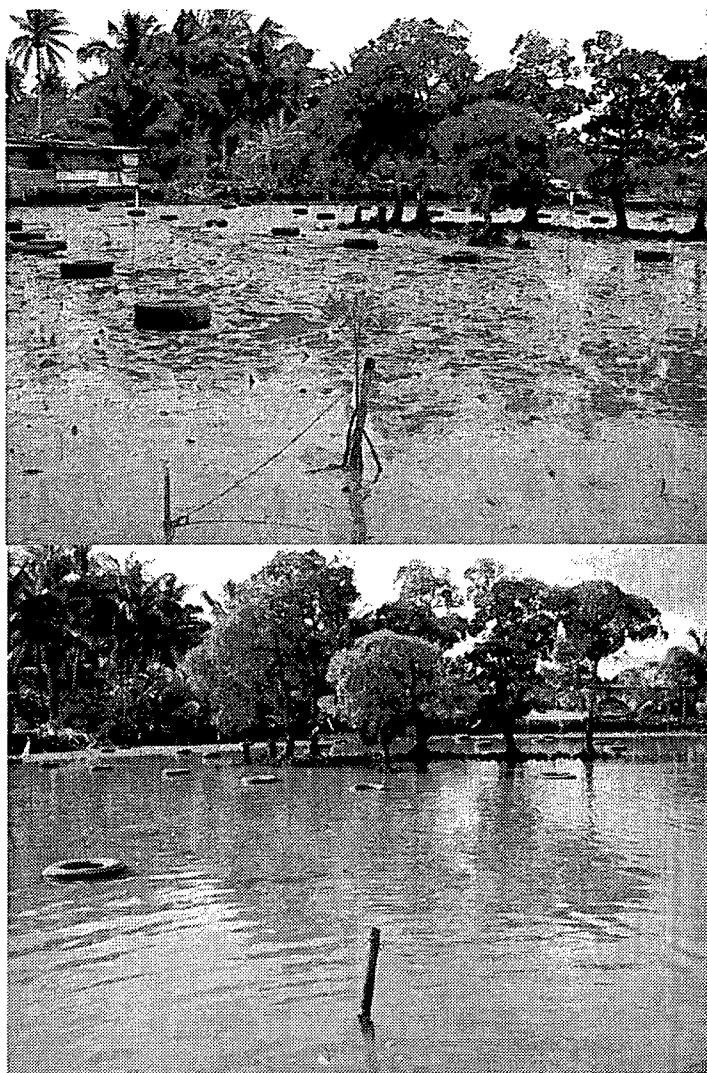


Fig. A1.6. Nu'uuli mangrove, American Samoa restoration study site from the Northern-most point looking South at time of completion of initial restoration activities, 15 June 2006 (top), and after six months (8 December 2006).

#### **A1.3.6. Monitoring and Maintenance**

The survival of each sapling in the study area was monitored five times over the six months following project initiation. For saplings planted inside of car tires, the height of sediment inside the tire relative to the sediment elevation of the substrate adjacent to the tire was measured at project initiation and at six months. Debris such as logs, garbage, timber and palm fronds were

periodically removed from the study site area. This was to prevent these rolling at high tide and dislodging saplings.

#### **A1.3.7. Project Costs**

Project costs are itemized in order to estimate a per hectare cost for mangrove restoration with the techniques employed in this study.

### **A1.4. RESULTS**

#### **A1.4.1. Cause of Mangrove Decline**

Figs. A1.7 – A1.9 show a time series of the study site at six points in time from 1961 through 2005. Analysis of these aerial photos and space imaging showed substantial loss of mangrove trees occurred between 1971 and 1984, with further reduction in area of a mangrove island between 1984 and 1990. The mangrove area margins and cover remained relatively unchanged between 1990 and 2004. In 1984 there was a substantial increase in development of adjacent upland areas relative to 1971 along the Coconut Point area near the study site.

Interviews with landowners adjacent to the mangrove restoration study site identified the cause of the original mangrove loss. In 1979, Leon and Michael Malau'ulu, then eight and ten years old, respectively, following their parent's instructions, spent one month cutting down the mangroves fronting their property using machetes (Fig. A1.10). The three reasons that Leon identified for their family's interest in removing the mangrove wetland were to (i) provide a source of firewood, (ii) improve boat access, and (iii) improve access to mudflat habitat for collecting a marine worm (lpo in Samoan) for subsistence consumption (personal communication, Leon Malau'ulu, 12 June 2006).



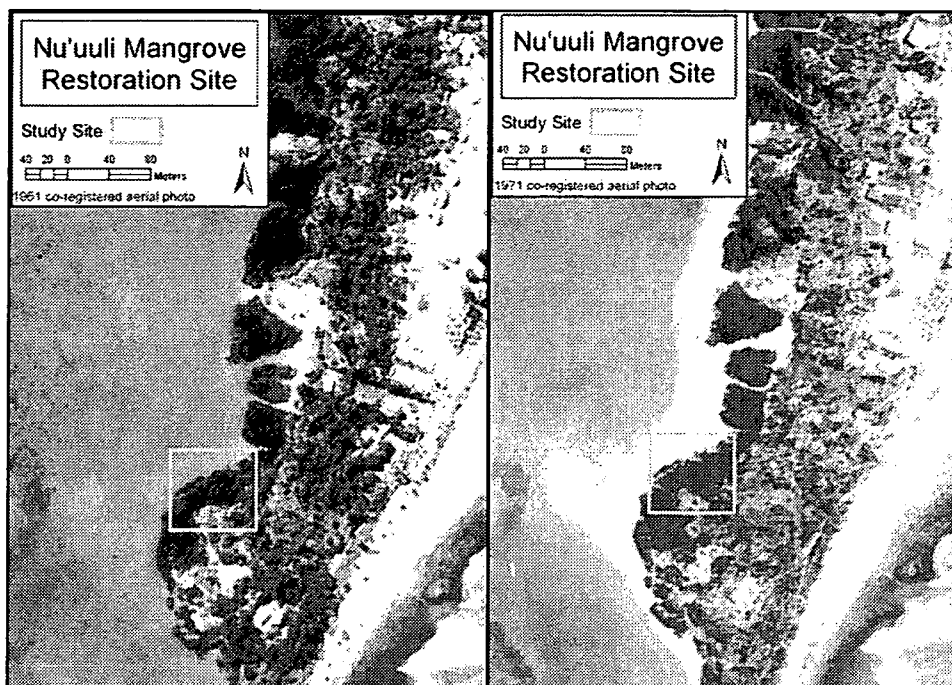


Fig. A1.7. Co-registered aerial photographs showing the mangrove restoration study site, 1961 (left) and 1971.

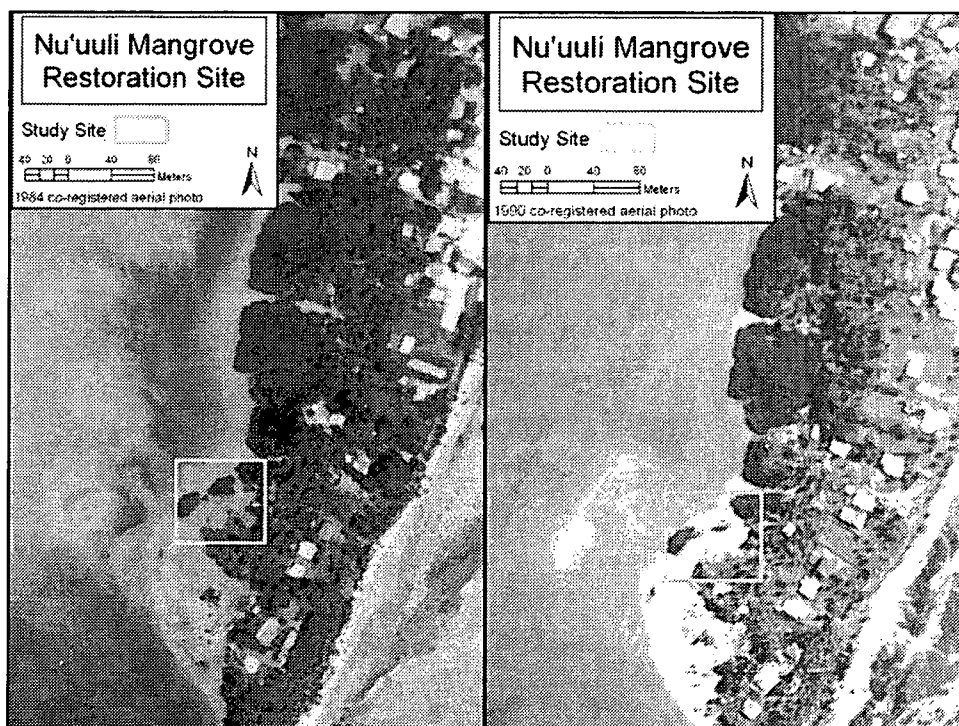


Fig. A1.8. Co-registered aerial photograph showing the mangrove restoration study site, 1984 (left) and 1990.

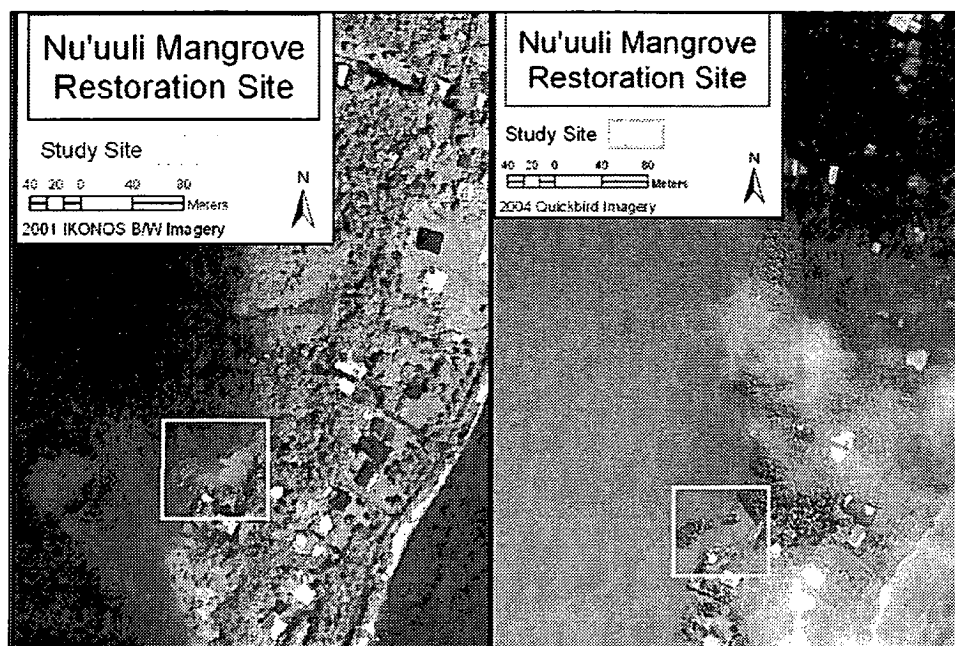


Fig. A1.9. 2001 IKONOS satellite imagery (left) and 2004 QuickBird satellite imagery showing the mangrove restoration study site.



Fig. A1.10. Mangrove stump, perhaps a remnant from the original 1979 clearing of the restoration study site.

Fig. A1.2 shows a covered structure on the shoreline where the landowner adjacent to the study site loads provisions onto small boats, which has likely contributed to preventing the long-term survival of mangrove seedlings that may have become established through natural recruitment since the Malau'ulu family cleared the mangrove trees from the area in 1979.

#### **A1.4.2. Community Participation**

The American Samoa Department of Commerce, Coastal Management Program obtained authorization and expression of support from the Nu'uuli mayor (Vaealuga Maae, Magele [High Talking Chief] of Nu'uuli village), Nu'uuli village council, and landowners adjacent to the study site to conduct the mangrove restoration project. Leon Malau'ulu, the landowner immediately adjacent to the study site, directly participated in restoration and monitoring activities (Fig. A1.11). Leon explained that his family's main interests to help restore the mangrove habitat fronting their property include: (i) reducing salt spray damage to their property; (ii) reducing debris from washing up to their property line; (iii) improving habitat conditions for mangrove crabs; and (iv) providing protection from storm energy and erosion (personal communication, Leon Malau'ulu, 12 June 2006).

Through consultation with the Malau'ulu family, a boat channel was included in the design of the mangrove restoration site (Fig. A1.5) to allow for continued boat access to the facility where the family loads provisions onto small boats, in part, to reduce the likelihood of future disturbance of the re-established mangrove wetland vegetation. The borders of the boat channel were lined with sufficiently tall rebar with brightly colored flag tape attached to the top ends to ensure visibility at high tide and discourage the boat operators from traveling into the mangrove area.



Fig. A1.11. Leon Malau'ulu, who in 1979 cleared the mangrove restoration area, and Soli Tuaumu, Wetlands Officer of the American

Samoa Coastal Management Program,  
conducting mangrove restoration activities, 12  
June 2006.

#### **A1.4.3. Width of Reference vs. Restoration Site Mangrove Vegetation Zones**

The width perpendicular to the shoreline of the reference site *B. gymnorrhiza* zone is 12.4 m, and width of the *R. mangle* zone is 19.1 m. The width of the restoration site *B. gymnorrhiza* zone is 19.7 m, and width of the *R. mangle* zone is 9.5 m. Due to difficulties encountered in locating suitable sources of *R. mangle* saplings to transplant to the restoration site, the *R. mangle* zone of the restoration site is narrower than in the reference site, and the *B. gymnorrhiza* zone is wider and extends further seaward than the restoration site.

#### **A1.4.4. Elevations of Reference Mangrove and Restoration Site**

The lowest elevation of the *R. mangle* reference mangrove site is used as the 0 mm elevation for this analysis. The seaward margin of the *R. mangle* zone is estimated to be at about mean sea level (Ellison, 2001, 2004). The *R. mangle* zone of the reference mangrove had a range in elevation of 0 - +332 mm, mean of 149 mm ( $\pm 40$  mm standard deviation of the mean, N = 7). The *B. gymnorrhiza* zone of the reference mangrove had a range in elevation of +384 - +393 mm, mean of 387 mm ( $\pm 3$  mm standard deviation of the mean, N = 5).

The *R. mangle* zone of the restoration site had a range in elevation of -46 - +94 mm, mean of 16 mm ( $\pm 6$  mm standard deviation of the mean, N = 26). The *B. gymnorrhiza* zone of the restoration site had a range in elevation of +43 - +250 mm, mean of 134 mm ( $\pm 12$  mm standard deviation of the mean, N = 27).

The initial elevations inside of tires in the restoration site *R. mangle* zone relative to the immediately adjacent substrate had a range in elevation of +98 - +240 mm, mean of 149 mm ( $\pm 10$  mm standard deviation of the mean, N = 17). The elevations inside of tires in the restoration site *B. gymnorrhiza* zone relative to the immediately adjacent substrate had a range

in elevation of +80 - +357 mm, mean of 143 mm ( $\pm$  9 mm standard deviation of the mean, N = 35).

#### **A1.4.5. Survival by Treatment, Species Zone and Area**

Fig. A1.12 shows the percent of saplings combined species that survived by date. At the last date of monitoring in December 2006, 38.0% of the original 137 saplings remained alive. Fig. A1.13 shows the percent of saplings by species that survived by date. After six months from project initiation, 45.3% and 21.4% of the original 95 *B. gymnorrhiza* saplings and 42 *R. mangle* saplings, respectively, remained alive. Fig. A1.14 shows a plan view of the study area identifying the locations of saplings that survived vs. died. Finally, Fig. A1.15 shows the percent of saplings by treatment (next to a rebar, inside a tire or with no support structure) that survived by date. After six months from project initiation, 36.8%, 34.6% and 52.9% of the original 68 saplings adjacent to rebar, 52 saplings inside a tire and 17 saplings with no support structure, respectively, remained alive.

For *R. mangle* saplings, 26.3% and 17.4% of those in tires (N = 19) and next to rebar (N = 23), respectively, remained alive after six months from project initiation. There were no *R. mangle* saplings associated with no support structure. For *B. gymnorrhiza* saplings, 39.4%, 46.7% and 52.9% of those in tires (N = 33), next to rebar (N = 45) and with no support structure (N = 17), respectively, remained alive after six months from project initiation.

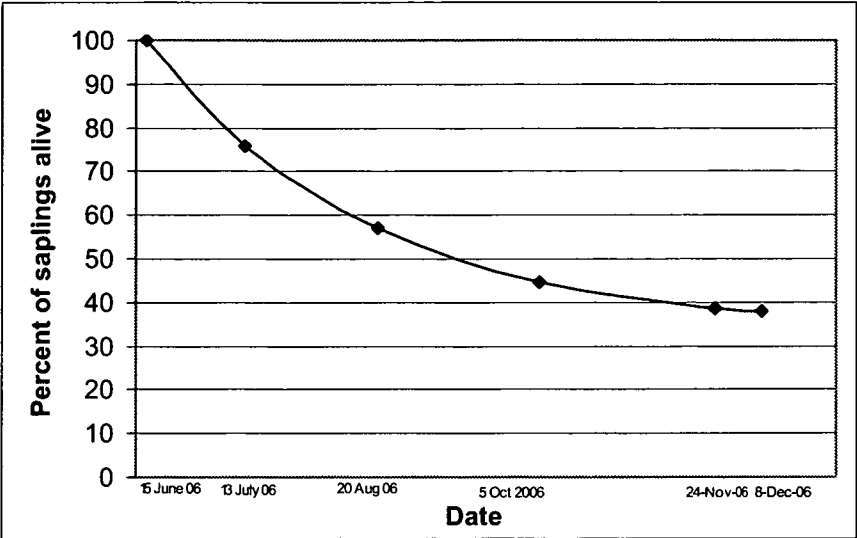


Fig. A1.12. Percent of surviving saplings (combined species, N = 137) vs. date, Nu'uuli mangrove restoration site.

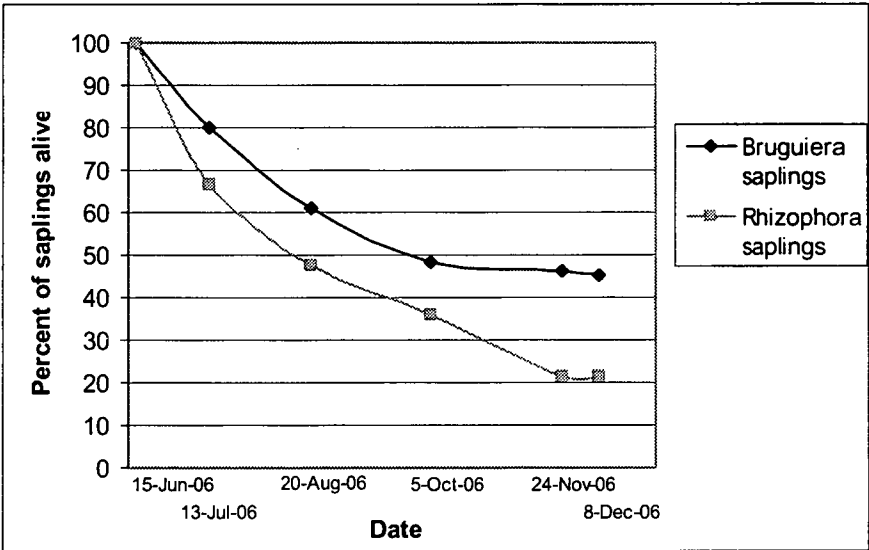


Fig. A1.13. Percent of surviving *B. gymnorhiza* saplings (N = 95) and *R. mangle* saplings (N = 42) vs. date, Nu'uuli mangrove restoration site.

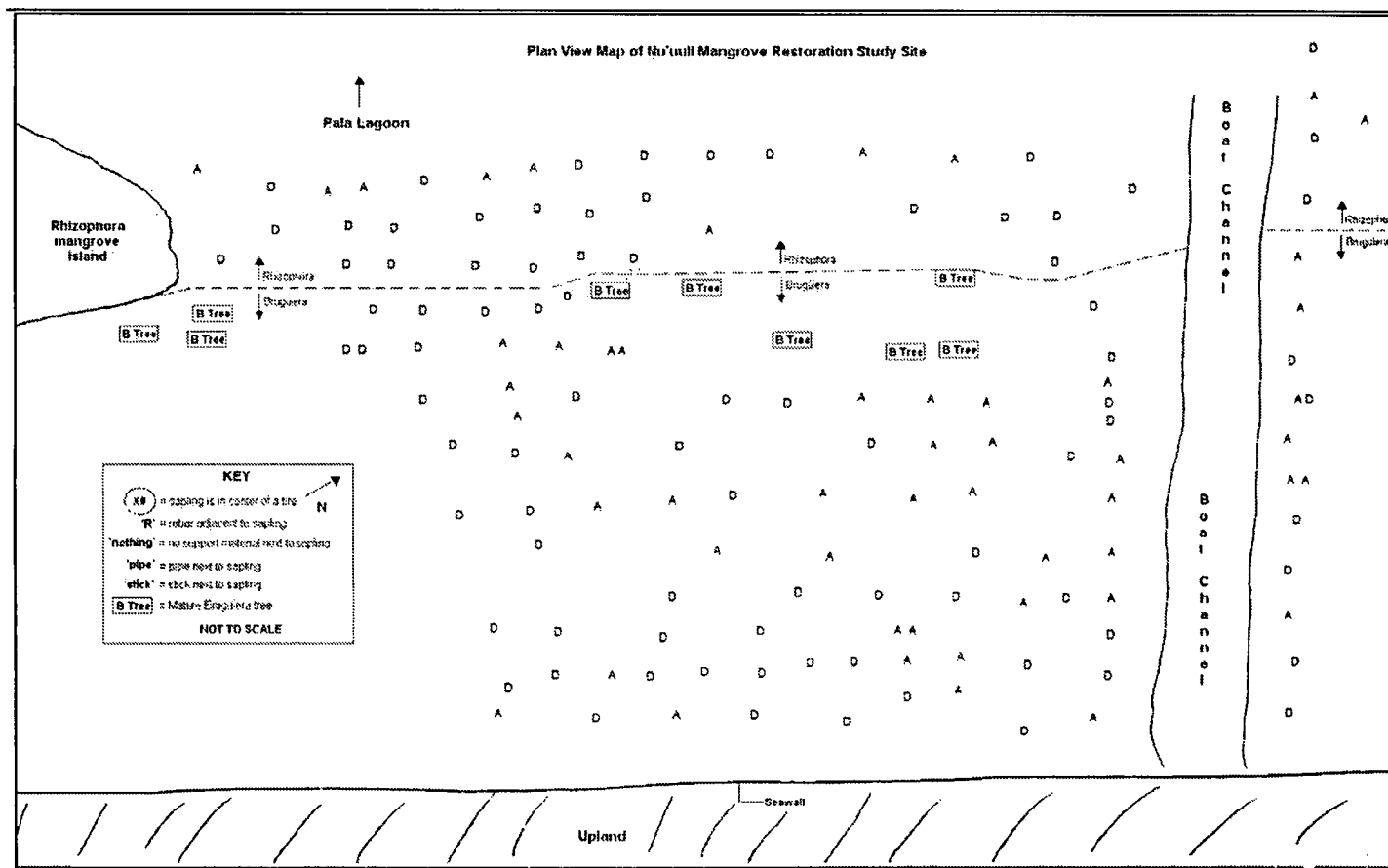


Fig. A1.14. Plan view of the Nu'uuli mangrove restoration study site identifying the location of saplings that survived vs. died from 15 June – 8 December 2006.

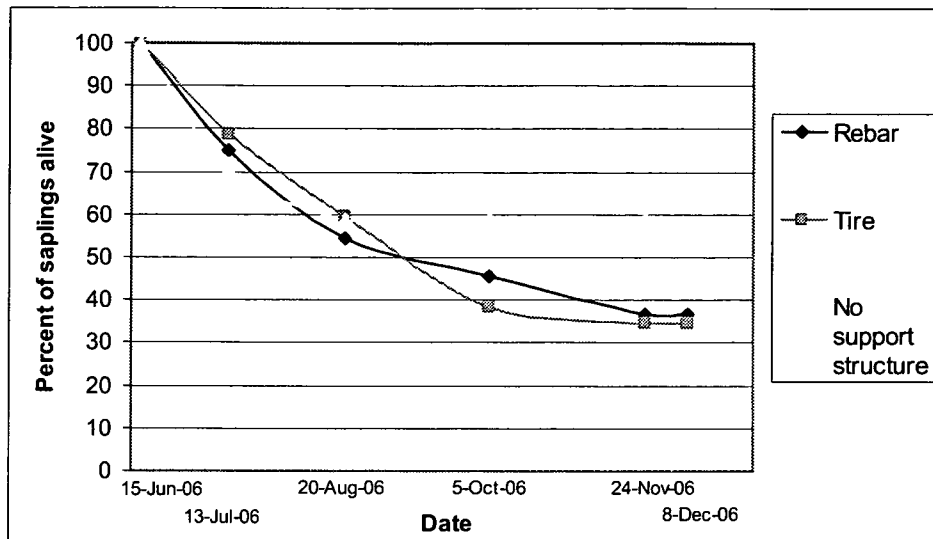


Fig. A1.15. Percent of saplings (combined species) by treatment (next to a rebar, N = 68, inside a tire, N = 52, or with no support structure, N = 17) that survived vs. date.

#### A1.4.6. Change in Height of Sediment Inside Tires

The elevation of sediment inside of tires into which mangrove saplings were planted lowered by an average of 46 mm from 15 June (average elevation above the ambient sediment surface of 138 mm,  $\pm$  4.9 mm standard deviation of the mean, N = 52) to 8 December 2006 (average elevation above ambient of 92 mm ( $\pm$  4.4 mm standard deviation of the mean, N = 52).

#### A1.4.7. Project Costs

Project costs through the completion of six months of monitoring for equipment (rebar, flag tape, paint), paid labor (estimated at USD 15 per hour for government employees and a government contractor), volunteer labor (estimated at USD 5 per hour), and miscellaneous expenses (use of computers, office supplies, internet, phone and a government vehicle) was USD 1,450. Predicted costs over the following six months for monitoring, debris removal and replacing dead saplings for government and volunteer labor is USD 700.



## A1.5. CONCLUSIONS

Damage to sapling roots during transplanting likely was a contributing cause of the observed mortality. Human disturbance may also have been a factor. Debris, such as logs and garbage, were observed in the site, and may have rolled with the tides and damaged saplings, contributing to a portion of the observed sapling mortality. Researchers qualitatively observed an increase in the number of seedlings becoming established over the six months from the initial installation of the rebar and tires, perhaps a result of reduced human traffic and increased protection from debris.

The restoration site's seaward *R. mangle* zone had a mean elevation deficit of 0.13 m relative to the *R. mangle* zone of a reference site. Saplings planted inside of tires in the *R. mangle* restoration site zone were an average of +149 mm above the adjacent surface outside of the tire and were at elevations similar to that in the *R. mangle* zone of the reference site. While sample sizes were small, the observed lower mortality of *R. mangle* saplings in tires vs. next to rebar suggests that the correction in elevation accomplished by the tires may have been a factor.

The restoration site's landward *B. gymnorrhiza* zone had a mean elevation deficit of 0.25 m relative to the *B. gymnorrhiza* zone of the reference site. Saplings in the *B. gymnorrhiza* restoration zone were an average of +143 mm above the adjacent surface outside of the tire and where an average of -110 mm below the elevation of the *B. gymnorrhiza* zone in the reference site. There was lower survival of *B. gymnorrhiza* saplings in tires vs. next to rebar or with no support structure. Again, while sample sizes were small, this suggests that factors other than differences in period, frequency and depth of inundation were predominant in causing observed survival rates in the *B. gymnorrhiza* mangrove zone. The transplanted saplings with no support structure fared better than those placed inside tires and next to rebar (Fig. A1.15). The small sample size of saplings with no support structure relative to the other two treatments, and predominant location in the landward portion of the restoration site where they may be relatively more protected from disturbance from debris (all saplings with no support are in the landward *B. gymnorrhiza* zone) may explain the higher survival of this treatment relative to the other two treatments. The seaward-most 7.3 m of the restoration site's *B. gymnorrhiza*

zone extends further seaward than in the adjacent reference area, which raised the concern that there would be relatively high mortality in this area of the *B. gymnorrhiza* zone of the restoration site. However, this was not the case. This suggests that this species is able to tolerate the higher period, frequency and depth of inundation that this area is subject to due to being at a lower elevation than the reference site.

Some possible problems with the use of tires as a support structure for mangrove restoration include: (a) the water temperature inside the tires was observed to be substantially higher than the ambient water temperature, which could stress the mangrove inside the tire; (b) the inundation period inside the tire is altered. For instance, Fig A1.6, bottom, shows that the water level inside the tire at the left side of the photograph is lower than outside the tire when the tide was rising; and (c) erosion of sediment from inside tires of about 46 mm over six months was observed. If this continues over coming years, this could result in stress and mortality. Only one of the 52 tires used in the experiment was observed to be partially dislodged, in this case, as a result of air caught inside the tire and perhaps due to erosion of sediment from inside the tire. Also, (d) the tires may move during a storm when there is high wave and current energy.

There was a substantially higher mortality rate of saplings in the *R. mangle* zone relative to in the *B. gymnorrhiza* zone (Fig. A1.13). This may be a result of the source of a large proportion of the *R. mangle* saplings being located at an area relatively far from the restoration site where soil and other environmental parameters are different. Half of the seaward-most row of *R. mangle* saplings survived, an area that presumably would encounter a relatively high degree of disturbance from debris, suggesting that damage from debris may not have been a large cause of the observed *R. mangle* mortality.

Project costs through one year from initial restoration activities are estimated to be USD 2,150, which is USD 13,030 ha<sup>-1</sup>. Labor comprised about 84% of the total costs. The range of reported costs for mangrove restoration is USD 225 to USD 216,000 ha<sup>-1</sup>, not including the cost of the land (Ramsar Secretariat, 2001; Lewis, 2005). Replicating the American Samoa restoration technique in less developed countries and territories would cost less due to lower labor costs.

It was critical to have direct community participation and support for the restoration project. In particular, without the approval of the adjacent landowner, there would have been a high risk of human disturbance to the restoration site. Attempts to establish restrictions on the use of resources in ways other than building on customary systems of management are not likely to be effective in the most parts of the Pacific Islands region (Gilman, 1997, 2002). Stakeholders will be more likely to comply with restrictions on their traditional resource use activities if they understand and support the restrictions, which can be accomplished through direct community participation in conservation activities such as mangrove restoration (Gilman, 1997, 2002).

While several years of monitoring will ultimately be required to determine the project's success, the restoration project has proven to be modestly successful with 38% sapling survival. Several years of monitoring will be necessary to determine whether this degraded mangrove with an ample supply of propagules that showed no natural regrowth, with only a small difference in hydrologic regime relative to a reference site, can be rehabilitated by reducing disturbance by people and debris, without modifying the site's physical structure to correct the hydrology. The restoration project is a model for the community-based and low-cost approaches to ecological restoration needed in the region. Pilot projects using similar techniques at the other 15 Pacific Island Countries and territories where mangroves are indigenous may be worth pursuing. The amount of time and expense required to rehabilitate damaged mangroves, as demonstrated in this study, supports efforts to avoid and minimize mangrove degradation.

## Appendix 2

### Metadata for Historical Aerial Photos and Satellite Imagery of American Samoa Mangrove Study Sites

Table A2.1. Metadata for historical aerial photos and satellite imagery of Masefau, Nu'uuli, and Leone mangroves, Tutuila Island, American Samoa used in the research studies included in this thesis. All photos were scanned at a resolution of 1200 dpi.

Mangrove Site Shown	File Name <sup>a</sup>	Frame	Flight Line	Scale	Resolution	Date of Image	Type	Location	Co-registration Total Control Point Error and Total Root Mean Square Error	Estimated Maximum Co-registration Error from IKONOS 2001 Reference Image (m)
1961										
Nu'uuli	1200nuuuli604	604	Index	1":2,000'	0.55 m	9/16/1961	B/W	Tafuna Airport	0.008	2
Nu'uuli	1200tafuna638	638	Index	1":2,000'	0.55 m	9/16/1961	B/W	Tafuna Airport	0.008	2
Masefau	1200masefau616	616	Index	1":2,000'	0.55 m	9/16/1961	B/W	Fagaitua/Masefau	0.01	30
Leone	1200leone648	648	Index	1":2,000'	0.55 m	9/16/1961	B/W	Leone	0.0297	35
1966										
Leone	1200leone016	16		1":2,000'	0.55 m	3/3/1966	B/W	Setapa/Leone	0.041	4
Leone	1200fagatele048	48		1":2,000'	0.55 m	3/3/1966	B/W	Fagatele/Leone	0.041	4
1971										
Masefau	1200masefau3-8	3 8 (1004)	Index	1":14,400'	0.55 m	7/11/1971	B/W	Masefau	0.0139	4

Leone	1200leone3-90	3 90	Index	1":14,400'	0.55 m	8/17/1971	B/W	Leone	0.0033	3
Nu'uuli	1200nuuuli3-30	3 30	Index	1":14,400'	0.55 m	7/11/1971	B/W	Nuuuli	0.0033	3
1984										
Leone	1200fagatel e96	8342-96		1":1,700'	0.55 m	11/7/1984	B/W	Fagatele, Leone, Malaeloa	No data	3
Nu'uuli	1200nuuuli 8342-19subreg	8342-19		1":1,700'	0.55 m	11/7/1984	B/W	Nuuuli, Tafuna	No data	2
1990										
Nu'uuli	1200nuuuli 8-103	103 8		Unknown	0.55 m	8/27/1990	Color	Tutuila	0.0042	1
Leone	1200leone2 54	254 17			0.55 m		Color	Tutuila	0.003	2
Masefau	1200masef au2-674	674 2		6":7,700'	0.55 m	9/18/1990	Color	Masefau	0.005	10
Leone	1200leone1 0-690	690 10		6":9,500'	0.55 m	9/27/1990	Color	Tutuila	0.003	2
1994										
Nu'uuli	1200nuuuli 3553	3553 30-100		1:30,000	0.55 m	6/7/1994	Color	Nuuuli	0.002	4
Leone	1200fagatel e3557.tif	3557 30-100		1:30,000	0.55 m	6/17/1994	Color	Leone/Malae loa/Fagatele Bay	0.0029	3
Leone	1200fagatel e3556.tif	3556 30-100		1:30,000	0.55 m	6/17/1994	Color	Leone/Iilili/Pavaia/Vaitogi Villages	0.0029	3
Leone	1200TutW3 651.tif	3651 30-101		1:30,000	0.55 m	6/30/1994	Color	Leone/Agugulu Villages	0.0029	3
Masefau	1200masef au3659subr	3659 30-101		1:30,000	0.55 m	6/30/1994	Color	Fagaitua/Masefau/Alofau	No data	2

Appendix 2. Metadata for Historical Aerial Photos and Satellite Imagery of American Samoa Mangrove Study Sites

eg				Villages					
2001									
Leone, Masefau, and Nu'uuli					Pixel resolutions: panchromatic c 1 m x 1 m; multispectral 4 m x 4 m; fused 1 m x 1 m	9/15/2001	Tutuila and Aunu'u Islands, American Samoa	NA	NA
	IKONOS space imaging	NA	NA	NA					
2003									
Masefau and Nu'uuli					Pixel resolutions: panchromatic c 0.63 m x 0.63 m; multispectral 2.52 m x 2.52 m; fused 0.63 m x 0.63 m	11/9/2003 Masefau; 12/15/2003 Nu'uuli	Space imaging Masefau and Nu'uuli	NA	NA
	QuickBird space imaging	NA	NA	NA					
2004									
Leone					Pixel resolutions: panchromatic c 0.63 m x 0.63 m; multispectral 2.52 m x 2.52 m; fused 0.63 m x 0.63 m	5/20/2004	Space imaging Leone	NA	NA
	QuickBird space imaging	NA	NA	NA					

<sup>a</sup> File names as used by the American Samoa Department of Commerce Coastal Management Program.

## **Appendix 3**

### **Questionnaire for Regional Capacity Survey to Assess Vulnerability and Adapt to Mangrove Responses to Sea Level Rise and Other Climate Change Effects (Chapter 6)**

The following questionnaire form was used to guide the collection of information from ten Pacific Island Countries and territories to assess the technical and institutional capacity to predict mangrove vulnerability to projected change in regional relative sea-level and capacity for adaptation to predicted mangrove responses to climate change.



UNIVERSITY OF TASMANIA



## Questionnaire to Assess Technical and Institutional Capacity of Pacific Island States and Territories to Assess and Manage Mangrove Response to Relative Sea-Level Change

### Project Background and Purpose

The University of Tasmania, Secretariat of the Pacific Regional Environment Programme (SPREP), and United Nations Environment Programme Regional Seas Programme, are collecting information on the capacity of Pacific Island Countries and territories to assess and adapt to mangrove response to climate change, including changes in local sea-level. Coastal ecosystems and shoreline development are vulnerable to relative sea-level rise and other climate change outcomes. Land-use planners can take steps to plan ahead to minimize losses of coastal habitats and damage to coastal development from shoreline responses to climate change. This project helps implement SPREP's *Regional Wetlands Action Plan for the Pacific Islands*, which identifies priority management, capacity building, and research and monitoring regional actions for mangroves.

Please help by responding to the following questions on capabilities to assess and manage mangrove response to rising sea-level – thank you!

Please direct questions and return questionnaires (email, fax, or mail) to the project manager, Eric Gilman, with the Hobart, Australia-based University of Tasmania School of Geography & Environmental Studies at:

Eric Gilman  
University of Tasmania  
2718 Napuaa Place  
Honolulu, HI 96822 USA  
Phone: +1.808.988.1976  
Fax: +1.808.988.1440  
egilman@utas.edu.au

1. Name: \_\_\_\_\_ Title: \_\_\_\_\_  
Agency/organization name: \_\_\_\_\_  
Mailing address: \_\_\_\_\_  
Phone: \_\_\_\_\_ Fax: \_\_\_\_\_  
Email: \_\_\_\_\_

2. (a) Which agencies/entities manage mangroves in your country/territory?  
(b) How many separate mangrove sites are in your country/territory?  
(c) What is the total area of mangroves?  
(d) Which islands contain mangroves?



(e) What are citations for documents with the most current information on mangrove area?  
Please can you provide a copy of this document?

(f) Has there been any work in your country/territory to assess how mangroves or other coastal habitats will respond to projected relative sea-level rise? For instance, has your country completed a *National Communication Report to the UNFCCC* or *National Adaptation Plan of Action* that addresses planning for mangrove response to changes in climate and sea level? ☐ Yes ☐ No

(g) If YES, please provide a citation for relevant reports and please provide a copy:

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3. (a) Is there a tide gauge in your country/territory? ☐ Yes ☐ No

(b) If yes, how many years of tide gauge data are available? \_\_\_\_\_

(c) Length of tide gauge record through December 2005: \_\_\_\_\_

(d) How far away is the closest mangrove site from the tide gauge? \_\_\_\_\_ km

(e) How far away is the furthest away mangrove site from the tide gauge? \_\_\_\_\_  
km

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4. (a) Have mangrove boundaries been delineated and mapped? ☐ Yes ☐ No

(b) If yes, what percent of the total mangrove area has had boundaries delineated?  
\_\_\_\_\_ %

(c) What was the most recent year when mangrove boundaries were delineated (list multiple years if delineations were conducted for different mangroves in different years)? \_\_\_\_\_

(d) What methods were employed to delineate mangrove boundaries (check appropriate box or boxes)?

☐ Survey equipment

☐ GPS

☐ Aerial photo interpretation

☐ Satellite imagery interpretation

☐ Other: \_\_\_\_\_

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5. (a) List the names of the islands that have mangroves that also have contour (topographic) maps available and provide the percentage of mangrove islands with topographic map coverage:

(b) What are the contour intervals?

(c) What height datum is used?

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(d) When were the contour maps produced?

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6. (a) Are maps available for the islands containing mangroves that identify the location of buildings, roads, and other coastal development? ☐ Yes ☐ No

(b) If YES, list the names of the islands containing mangroves that have these maps available:

(c) When were these maps produced?

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7. (a) Is historical imagery available for coastlines containing mangroves? ☐ Yes ☐ No

(b) If YES, what types of imagery are available (e.g., aerial photos, maps, satellite imagery):

(c) What is the earliest date of available imagery? \_\_\_\_\_

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8. Have sedimentation rates or rate of change in elevation of the mangrove surface been measured in mangroves? ☐ Yes ☐ No

If YES, please provide a citation for a document reporting the results and please provide a copy.

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9. Is there monitoring of mangrove ecological parameters? ☐ Yes ☐ No

If YES, what parameters are monitored (e.g., salinity, pH, DBH, sedimentation, canopy cover, bird abundance)?

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10. Are there staff with skills to conduct mangrove surveys and inventories? ☐ Yes ☐ No

If YES, how many staff have the skills to conduct mangrove surveys? \_\_\_\_\_

What agencies or organizations do they work for?

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11. Have mangroves been rehabilitated? ☐ Yes ☐ No

If YES, what method was used (e.g., restore hydrology, restore elevation, planting):

If YES, what area of mangroves was attempted to be rehabilitated \_\_\_\_\_ and was successfully rehabilitated? \_\_\_\_\_

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12. Is there a monitoring program for habitats other than mangroves (check relevant boxes):

- ☐ None
- ☐ Coral reefs
- ☐ Salt marshes
- ☐ Sea grass beds
- ☐ Freshwater wetlands
- ☐ Lakes
- ☐ Streams
- ☐ Mudflats
- ☐ Beaches
- ☐ Upland forests
- ☐ Other \_\_\_\_\_

If YES, what parameters are monitored?

---

13. If there is a permit programme for coastal development, which agency manages the programme?

Is there a zoning programme to regulate where development is located? ☐ Yes ☐ No

Have site-specific mangrove vulnerability assessments been conducted? ☐ Yes ☐ No

If yes, please provide a reference to the report containing the results of the assessment(s).

Is there a plan for adaptation to coastal ecosystem responses to climate change? ☐ Yes  
☐ No

If yes, please provide a reference to the plan.

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14. Would your agency be interested in participating in a regional network to monitor mangroves and assess mangrove response to relative sea-level rise, if such a network is established?

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